UNITED STATES DEPARTMENT OF COMMERCE
National Oceanic and Atmospheric Administration
NATIONAL MARINE FISHERIES SERVICE
Northwest Region
7600 Sand Point Way N.E., Bldg. 1
Seattle, Washington 98115

Reply To:
NMFS Tracking No.:
2009/03531

January 30, 2012

Michael Bussel
Director, Office of Water and Watersheds
Environmental Protection Agency
1200 Sixth Avenue, Suite 900
Seattle, WA 98101-3140

Re: Issuance of an Incidental Take Permit, Endangered Species Act Section 7 Formal Consultation and Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat Consultation for Reissuance of the Fort Lewis (Joint Base Lewis McChord) Wastewater Treatment Facility NPDES Permit (WA-002195-4) (Sixth Field HUC# 171100190601, Chambers Creek) Pierce County, Washington.

Dear Mr. Bussel:

The enclosed document contains a biological opinion prepared by the National Marine Fisheries Service (NMFS) Habitat Conservation and Protected Resources divisions, pursuant to section 7(a)(2) of the Endangered Species Act (ESA). It analyzes the effects of the Enviromental Protection Agency (EPA's) proposal to authorize the NPDES permit on threatened and endangered fishes and marine mammals potentially affected by the proposed action. In this opinion, NMFS concludes that the proposed action is not likely to jeopardize the continued existence of Puget Sound Chinook, Puget Sound steelhead, Puget Sound/Georgia Basin yelloweye, canary and bocaccio rockfish, or Southern Resident killer whale.

As required by section 7 of the Endangered Species Act, the Services provided an incidental take statement with the biological opinion. The incidental take statement describes reasonable and prudent measures National Marine Fisheries Service considers necessary or appropriate to minimize incidental take associated with this action. The take statement sets forth nondiscretionary terms and conditions, including reporting requirements, that the Federal agency and any person who performs the action must comply with to carry out the reasonable and prudent measures. Incidental take from actions that meet these terms and conditions will be exempt from the Endangered Species Act take prohibition.

This document also includes the results of our analysis of the action's likely effects on essential fish habitat pursuant to Section 305(b) of the Magnuson-Stevens Fishery Conservation and Management Act (MSA), and includes six conservation recommendations to avoid, minimize, or otherwise offset potential adverse effects on essential fish habitat.

These Conservation Recommendations are a non-identical set of the Endangered Species Act Terms and Conditions. Section 305(b) (4) (B) of the MSA requires Federal agencies to provide a detailed written response to National Marine Fisheries Service within 30 days after receiving these recommendations.

If the response is inconsistent with the essential fish habitat conservation recommendations, the EPA must explain why the recommendations will not be followed, including the justification for any disagreements over the effects of the action and the recommendations. In response to increased oversight of overall essential fish habitat program effectiveness by the Office of Management and Budget, National Marine Fisheries Service established a quarterly reporting requirement to determine how many conservation recommendations are provided as part of each essential fish habitat consultation and how many are adopted by the action agency. Therefore, in your statutory reply to the essential fish habitat portion of this consultation, we ask that you clearly identify the number of conservation recommendations accepted.

If you have any questions, please contact Tim Rymer of my staff at the Washington State Habitat Office at (360) 753-4126, by e-mail at Tim.Rymer@noaa.gov, or by mail at the letterhead address.

Sincerely,

Michael Tehan

for William W. Stelle, Jr.
Regional Administrator

Enclosure

cc: Hanh Shaw

Endangered Species Act (ESA) Section 7(a)(2) Biological Opinion and Section 7(a)(2) "Not Likely to Adversely Affect" Determination
AND
Magnuson-Stevens Fishery Conservation and Management Act Essential Fish Habitat (EFH) Consultation

Reissuance of the Fort Lewis (Joint Base Lewis McChord) Wastewater Treatment Facility NPDES Permit (WA-002195-4) (Sixth Field HUC# 171100190601, Chambers Creek)
Pierce County, Washington.

NMFS Consultation Number: 2009/03531

Action Agency: U.S. Environmental Protection Agency

Affected Species and Determinations:

ESA-Listed Species	Status	Is Action Likely to Adversely Affect Species or Critical Habitat?	Is Action Likely To Jeopardize the Species?	Is Action Likely To Destroy or Adversely Modify Critical Habitat?
Puget Sound Chinook salmon (*Oncorhynchus tshawytscha*)	Threatened	Yes	No	No
Puget Sound steelhead (O. mykiss)	Threatened	Yes	No	Not Designated
Yelloweye rockfish (*Sebastes ruberrimus*)	Threatened	Yes	No	Not Designated
Canary rockfish (S. pinniger)	Threatened	Yes	No	Not Designated
Bocaccio (S. paucispinis)	Endangered	Yes	No	Not Designated
Southern Resident killer whale (*Orcinus orca)*	Endangered	Yes	No	No
Humpback whales (Megaptera novaeangliae)	Endangered	No	No	Not Designated
Steller sea lion (*Eumetopias jubatus*)	Threatened	No	No	Not Applicable

Fishery Management Plan That Describes EFH in the Project Area	Does Action Have an Adverse Effect on EFH?	Are EFH Conservation Recommendations Provided?
Pacific Coast Salmon	Yes	Yes

Consultation Conducted By: National Marine Fisheries Service, Northwest Region

Issued By:

for William W. Stelle, Jr.
Regional Administrator

Date: January 30, 2012

TABLE OF CONTENTS

LIST OF ACRONYMS AND ABBREVIATIONS

BE	Biological Evaluation
BRT	Biological Review Team
CFR	Code of Federal Regulations
cfs	Cubic Feet per Second
CH	Critical Habitat
CIG	Climate Impacts Group
COE	U.S. Army Corps of Engineers
CSO	Combined Sewage Overflow
cy	Cubic Yards
DBH	Diameter at Breast Height
DDT	Dichlorodi-phenyl trichloroethane
DFO	Canada Department of Fisheries and Oceans
DPS	Distinct Population Segment
DQA	Data Quality Act
EFH	Essential Fish Habitat
EPA	Environmental Protection Agency
ESA	Endangered Species Act
ESU	Evolutionarily Significant Unit
FIR	Food Intake Rate
FMR	Field Metabolic Rate
FR	Federal Register
HUC	Hydrologic Unit Code
IPCC	Intergovernmental Panel on Climate Change
ISAB	Independent Scientific Advisory Board
ITS	Incidental Take Statement
JBLM	Joint Base Lewis McChord
LWM	Large Woody Material
mgd	Million Gallons per Day
MSA	Magnuson-Stevens Fishery Conservation and Management Act
NMFS	National Marine Fisheries Service
NOEC	No Observed Effect Concentration
NPDES	National Pollution Discharge Elimination System
NWFSC	Northwest Fisheries Science Center
OHWL	Ordinary High Water Line
Opinion	Biological Opinion
PAH	Polycyclic Aromatic Hydrocarbon
PBDE	Polybrominated Diphenyl Ether
PBT	Persistent Bioaccumulative Toxicant
PCE	Primary Constituent Element
PFMC	Pacific Fishery Management Council
PL	Public Law
POP	Persistent Organic Pollutant

POTW	publicly owned treatment works
PPCP	pharmaceuticals and personal care products
PSAMP	Puget Sound Assessment and Monitoring Program
PSSTRT	Puget Sound Steelhead Technical Recovery Team
PSTRT	Puget Sound Technical Recovery Team
RIP	Rehabilitation and Inspection Program
RM	River Mile
RPMs	Reasonable and Prudent Measures
TMDL	Total Maximum Daily Load
U.S.C.	United States Code
VSP	Viable Salmonid Population
WDFW	Washington Department of Fish and Wildlife
WDOE	Washington Department of Ecology
WET	Whole Effluent Toxicity Testing
WWTP	Waste Water Treatment Plant

1. INTRODUCTION

This Introduction section provides information relevant to the other sections of this document and is incorporated by reference into Sections 2 and 3 below.

1.1 Background

The biological opinion (opinion) and incidental take statement portions of this document were prepared by the National Marine Fisheries Service (NMFS) in accordance with section 7(b) of the Endangered Species Act (ESA) of 1973, as amended (16 U.S.C. 1531, et seq.), and implementing regulations at 50 CFR 402.

The NMFS also completed an Essential Fish Habitat (EFH) consultation. It was prepared in accordance with section 305(b)(2) of the Magnuson-Stevens Fishery Conservation and Management Act (MSA) (16 U.S.C. 1801, et seq.) and implementing regulations at 50 CFR 600.

The opinion and EFH conservation recommendations are both in compliance with the Data Quality Act (44 U.S.C. 3504(d)(1) et seq.) and they underwent pre-dissemination review.

The Fort Lewis (Joint Base Lewis McChord) Wastewater Treatment Plant (WWTP) at Solo Point is a federally-owned, trickling filter plant that provides secondary treatment and disinfection of domestic wastewater and limited industrial wastes. McChord Air Force Base was recently combined with the Fort Lewis Army Training facility under a base realignment and restructuring program. Joint Base Lewis McChord (JBLM) provides services to a military and civilian workforce of 29,000 people. This facility treats wastewater from Fort Lewis Army Training Center, McChord Air Force Base, Madigan Army Medical Center, the Veteran's Hospital at American Lake, and Camp Murray National Guard Station. The maximum and average design flow rates of the WWTP are 15 million gallons per day (mgd) and 7 mgd, respectively. Currently the average daily flow is 3.77 mgd. The last National Pollution Discharge Elimination System (NPDES) permit for this facility was issued on February 1, 2004 and expired February 1, 2009. A timely and complete application for renewal of the permit was submitted by JBLM. Therefore the 2004 permit has been continued under 40 CFR 122.6 and remains fully effective and enforceable until it can be reissued.

In 2009, Fort Lewis supported approximately 104,000 civilian and military personnel. The Department of the Army is proposing a Fort Lewis Army Growth and Force Structure Realignment that will increase the number of supported personnel to 118,400 over the next 6 years. This will result in a greater volume of effluent being discharged at the Solo Point WWTP. Also, in order to accommodate the multiple needs of this growing population it will necessitate new roads, infrastructure and housing. The increase in impervious surface associated with development will result in greater amounts of surface-water runoff, sediment loads and contaminants into the action area over time. Population growth in the areas surrounding JBLM will occur and increase peak stormwater discharge into Chinook salmon critical habitat and Southern Resident killer whale critical habitat by off-site facilities.

1.2 Consultation History

The following chronology documents key points of the consultation process that led to this opinion for Puget Sound Chinook salmon, their designated CH, Puget Sound steelhead, SR killer whales, their designated CH, and listed rockfish.

On June 29, 2009, NMFS received a draft Biological Evaluation (BE) and request from the United States Environmental Protection Agency (EPA) for informal consultation under ESA section 7. The EPA proposed to reissue the National Pollution Discharge Elimination System (NPDES) permit for the Solo Point facility at Joint Base Lewis McChord (JBLM). The point of discharge is located offshore at Solo Point in Puget Sound, Sixth Field Hydrologic Unit Code (HUC 6) 171100190601. The EPA determined the proposed action: (1) is not likely to adversely affect Puget Sound Chinook salmon (*Oncorhynchus tshawytscha*), Puget Sound steelhead (*O. mykiss*), and Southern Resident (SR) killer whales (*Orcinus orca*); (2) will not adversely affect designated critical habitat (CH) for Puget Sound Chinook salmon and SR killer whales; and (3) will not affect designated EFH utilized by Pacific salmon, groundfish, and coastal pelagic species.

On August 6, 2009, NMFS discussed concerns associated with potential effects of the action with the EPA project manager, Tonya Lane.

On February 10, 2010, NMFS mailed a non-concurrence letter to EPA related to their determination of effects. The NMFS did not concur with the EPA's determinations. After reviewing effects of the action, NMFS determined that the actions are likely to adversely affect Puget Sound Chinook salmon, their designated CH, and Puget Sound steelhead. Critical habitat has not been designated for Puget Sound steelhead. The NMFS also determined the proposed action will adversely affect designated EFH. Included was a request for additional information necessary to initiate formal consultation. Given the action would occur after July 27, 2010, the effective listing date for Puget Sound/Georgia Strait Bocaccio (*Sebastes paucispini)*, Puget Sound/Georgia Basin yelloweye rockfish (*S ruberrimus*), and Puget Sound/Georgia Basin canary rockfish (*S. pinniger*), NMFS suggested the EPA include these species in the consultation. Critical habitat has not been designated for Puget Sound/Georgia Basin bocaccio, Puget Sound/Georgia Basin yelloweye rockfish, or Puget Sound/Georgia Basin canary rockfish.

On June 4, 2010, NMFS received a request for formal consultation and an addendum to the draft BE that was provided on June 29, 2009, from EPA.

On July 19, 2010, NMFS received from EPA a second BE Addendum that was prepared to address the listing of the rockfish which had occurred subsequent to the original request for consultation. Formal consultation was initiated at this time.

On August 4, 2010, NMFS and EPA met to discuss the action and status of the consultation.

On May 16, June 9, July 26, August 25, and September 12, 2011, NMFS and EPA met to discuss formal consultation for Southern Resident killer whales based on anticipated effects to Southern

Resident killer whales and conservation measures for killer whales. During these meetings the agencies also discussed anticipated terms and conditions for salmon.

These exchanges regarding potential effects on species resulted in an expansive biological review. The full list of species addressed in this consultation appears in Table 1, which also includes Federal Register notices for final rules that list threatened and endangered species designate critical habitat, or apply protective regulations.

Table 1: Federal Register notices for final rules that list threatened and endangered species, designate CHs, or apply protective regulations to listed species considered in this consultation.

Species	ESU or DPS	Original Listing Notice	Listing Status Reaffirmed	Critical Habitat	Protective Regulations
Chinook salmon *(Oncorhynchus tshawytscha)*	Puget Sound	3/24/99 64 FR 14308	8/15/11 76FR50448 Threatened	9/02/05 70 FR 52630	6/28/05 70 FR 37160
Steelhead *(O. mykiss)*	Puget Sound	5/11/07 72 FR 26722 Threatened	8/15/11 76FR50448 Threatened	In development	9/25/08 73 FR 55451
Killer whale (*Orcinus orca)*	Southern Resident	11/18/2005 70 FR 69903 Endangered	Not applicable	11/29/2006 71 FR 69054	ESA section 9 applies
Yelloweye rockfish (*Sebastes ruberrimus*)	Puget Sound/ Georgia Basin	4/28/2010 75 FR 22276 Threatened	Not applicable	In development	In development
Canary rockfish (*S. pinniger*)	Puget Sound/ Georgia Basin	4/28/2010 75 FR 22276 Threatened	Not applicable	In development	In development
Bocaccio *(S. paucispinis)*	Puget Sound/ Georgia Basin	4/28/2010 75 FR 22276 Endangered	Not applicable	In development	ESA section 9 applies

On October 10, 2011, EPA shared with NMFS their conservation measures for the species listed under the ESA that are addressed in this opinion..

A complete record of this consultation is on file at the Washington State Habitat Office in Lacey, Washington.

1.3 Proposed Action

"Action" means all activities or programs of any kind authorized, funded, or carried out, in whole or in part, by Federal agencies. Interrelated actions are those that are part of a larger action and depend on the larger action for their justification. Interdependent actions are those that have no independent utility apart from the action under consideration.

The EPA proposes to reissue a NPDES permit for the JBLM WWTP located at Solo Point in Pierce County, Washington. The NPDES permitting program is authorized by Section 402 of the Clean Water Act and implemented by regulations appearing in Part 122 of Title 40 Code of Federal Regulations (CFR) as well as other Parts of 40 CFR. Issuing the permit enables the continuation of ongoing disposal of domestic and industrial wastewater in compliance with the Clean Water Act for the next five-year period.

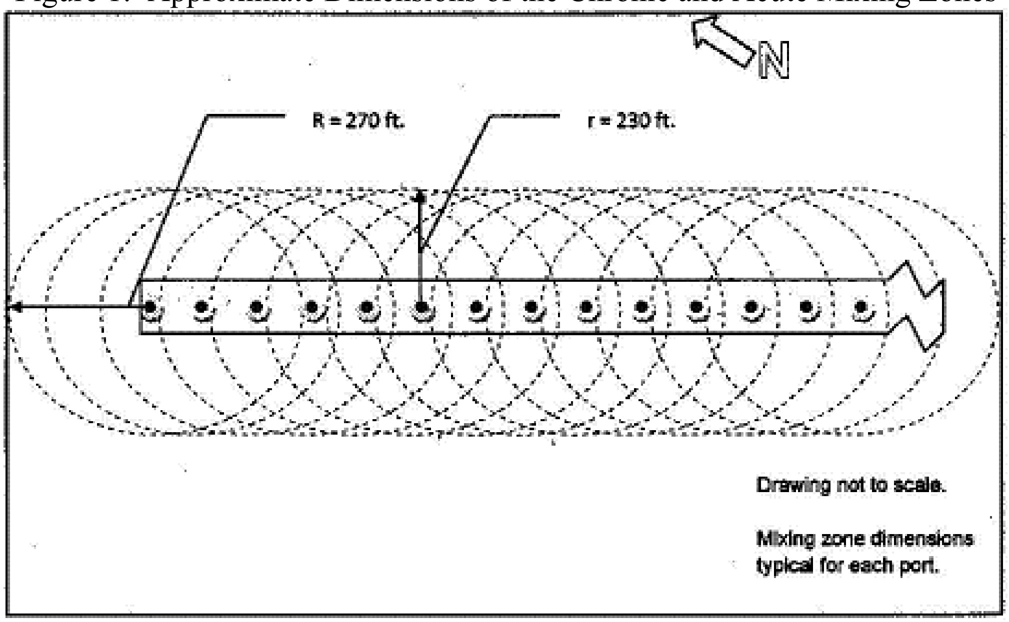
The draft permit for this action proposes effluent limits for: 1) 5-day biochemical oxygen demand with an average monthly limit of 30 milligrams per liter (mg/L) or 1,751 pounds per day and an average weekly limit of 45 mg/L or 2,627 pounds per day; 2) Total suspended solids with an average monthly limit of 30 mg/L or 1,751 pounds per day and an average weekly limit of 45 mg/L or 2,627 pounds per day; 3) Fecal coli form bacteria with an average monthly limit of 200/100 milliliters (mL) and an average weekly limit of 400/100 mL; 4) Total residual chlorine with an average monthly limit of 0.36 mg/L and a maximum daily limit of 0.50 mg/L; 5) Total petroleum hydrocarbon with a maximum daily limit of 10 mg/L; and 6) pH of 6.0-8.5 standard units.

The following parameters have no current effluent limits but will be monitored during the new permit cycle: total ammonia, total Kjeldahl nitrogen, nitrogen, total phosphorus, temperature, NPDES Application Form 2A Effluent Testing constituents, whole effluent toxicity testing and pretreatment parameters that include priority pollutant metals (arsenic, cadmium, chromium, copper, lead, mercury, nickel, selenium, silver, zinc and cyanide) and the priority pollutant organics (see Appendix A). Phthalates were identified as being present in the discharge as part of the NPDES permit renewal application. Also, upon request for additional information of general pollutants in the effluent stream, EPA provided detail on pharmaceuticals and personal care products (PPCPs) as well as flame retardants (PBDEs).

The existing 24-inch diameter outfall discharges effluent to Puget Sound through a 130-foot long, 14 port diffuser. Each of the 6-inch diameter ports is separated by a distance of 10 feet. The outfall extends from its closest point, approximately 370 feet from shore, to 500 feet at its furthest point. The diffuser depth at mean lower low water ranges from 70 feet at its deepest point, and shallower. There are no plans to modify this structure at this time.

With this new NPDES permit, the EPA anticipates discharges that exceed state water quality standards within a "mixing zone." In WAC 173-201A-020, a "Mixing Zone" is defined as that portion of a water body adjacent to an effluent outfall where mixing results in the dilution of the effluent with the receiving water. Water quality criteria may be exceeded in a mixing zone as conditioned and provided for in WAC 173-201A-400. The Washington State Department of Ecology at WAC 173-201A-400(7)(b) specifies that mixing zones must not extend in any horizontal direction from the discharge ports for a distance greater than 200 feet plus the depth of water over the discharge ports as measured during mean lower low water (MLLW). In this instance that distance is 70 feet. Therefore, the proposed chronic mixing zone under the new permit will be 270 feet long (a reduction from 300 feet under the existing permit) at each end of the diffuser (130 + 270 + 270) for a total of 670 feet. The width of the mixing zone of 460 feet will remain unchanged under the new permit. The total area of the chronic mixing zone, shown in Figure 1 (within the dashed lines) is approximately equal to 308,200 square feet, or about 7 acres in size, and extends from the seabed to the water surface. The acute mixing zone is identified as one tenth of the distance to the boundaries of the chronic mixing zone in all directions. Figure 1 depicts the approximate size and location of the outfall and its associated chronic and acute mixing zones.

Figure 1: Approximate Dimensions of the Chronic and Acute Mixing Zones



The EPA has also identified conservation measures that they included as part of this proposed action. Conservation measures represent actions that the action agency or the applicant will implement to further the recovery of the listed species and may include tasks recommended in the species recovery plan. Because the conservation measures are part of the proposed action under consultation, and their implementation is evaluated as part of the effects analysis, and thus become required components under the terms of this consultation. In this instance, EPA and its applicant, the DOD, commit to:

1) Implementation of a pretreatment program that monitors influent, effluent, and sludge for parameters confirmed present following priority pollutant effluent scans, as well as total petroleum hydrocarbon and other parameters of particular concern.

2) Influent and effluent monitoring for polybrominated diphenyl ethers (PBDEs) during the permit cycle. The proposed monitoring would consist of 8 effluent (and 8 concurrent influent) grab samples obtained during quarterly sampling the first full year and in the final year of the permit cycle.

3) The institution of a local sewer use ordinance.

4) Submittal of a sewage sludge permit within six months of the new permit effective date.

5) Updating a 1998 mixing zone study to more accurately determine the dilution available to the discharge.

6) Obtaining grab samples outside the mixing zone to evaluate if the effluent is causing or contributing to an exceedence of water quality criteria.

7) Conducting an outfall inspection to evaluate the physical condition of the discharge pipe and diffusers, and evaluate the extent of sediment accumulation in the vicinity of the outfall.

8) Quantitative survey of the habitat, to be conducted during the outfall inspection.

9) Continued collection system inflow and infiltration repair and reporting. This includes a requirement to prepare a feasibility study and engineering report to demonstrate a commitment to upgrade this aging treatment facility.

10) Direct reporting. EPA will require the JBLM to send NMFS directly the habitat survey report, and annual reports.

11) Pursuit of funding. The JBLM will send a letter to EPA's regional administrator committing to pursuing funds necessary for plant upgrades and the general timeframe for plant construction and operation.

12) Technology upgrades. EPA will include a statement asking that JBLM consider treatment technologies for PBDEs as the base plans for an upgrade to Solo Point.

13) Cooperative participation in a Policy Forum with Washington Department of Ecology and NMFS to address PBDE discharges from the major Puget Sound municipal treatment plants. EPA will convene a meeting(s) with senior management and technical staff from the respective agencies to present available information on: 1) the risk that PBDE municipal plant discharges have on Killer Whales, 2) opportunities to require PBDE monitoring in NPDES permits, and 3) strategies to reduce Puget Sound PBDE loadings from municipal treatment plants. Milestones for this measure include:

a. The EPA will schedule a half-day meeting to discuss this topic by no later than August 31, 2012.
b. EPA will write a letter to Ecology not later than October 1, 2012 recommending that PDBE influent/effluent monitoring be conducted at identified municipal treatment plants and included as a requirement in re-issued NPDES permits.
c. In the 2012-2017 timeframe, EPA and NMFS will coordinate with Ecology on the re-issuance of one Ecology-issued NPDES permit to address PBDEs unless EPA and NMFS determine the issues are being adequately addressed.

14) Establishment of a Technical Workgroup: EPA will work with NMFS to establish and convene a workgroup that will seek opportunities to fill data gaps and refine uncertainties about the effects of PBDEs on Southern Resident killer whales and wastewater treatment technologies to minimize those effects. NMFS and EPA anticipate that the workgroup will be comprised of scientists employed by NMFS, EPA, and other agencies involved in the study and management of contaminants and marine mammals, including the Washington Department of Ecology, the Puget Sound Partnership, and the Washington Department of Fish and Wildlife. The following schedule of activities and milestones shall apply to the workgroup:

a. The NMFS and EPA will identify individuals for the workgroup and hold the first meeting or conference call by April 3, 2012. Prior to the meeting or call, the workgroup will be provided a copy of the signed biological opinion and supporting documentation of the Southern Resident killer whale effects analysis and a list of topics to address as listed in b. below.
b. The following is the initial list of topics for consideration by the workgroup to reduce uncertainties related to PBDE effects on Killer Whales associated with the treatment plant discharges into the Puget Sound and the options for minimizing those effects: Technology Review of PBDE Removal Effectiveness in Wastewater Treatment Plants; PBDE Modeling in Puget Sound; Toxicity Reference Values for PBDEs in Marine Mammals; No Observed Effect Concentration (NOEC) Levels for Mixtures of PBDE and Polychlorinated Biphenyl (PCB) Congeners.
c. By June 30, 2012, the workgroup will identify opportunities, project ideas, and available funding sources, to address the topics listed above.
d. By October 31, 2012, the workgroup will support agencies/entities to carry-out work efforts or develop project proposals to address the topics listed above. The timeframe for this milestone may be adjusted to meet request for proposal or potential funding opportunity timelines and may be different for different projects.
e. By no later than November 30, 2012, EPA and NMFS will provide a status summary of the workgroup recommendations and efforts to the policy forum identified under conservation measure 13.

Figure 2: Approximate size and location of the Acute and Chronic Mixing Zones

1.4 Action Area

"Action area" means all areas to be affected directly or indirectly by the Federal action and not merely the immediate area involved in the action (50 CFR 402.02). For this consultation, the action area for fish includes the acute and chronic mixing zones surrounding the Solo Point outfall in South Puget Sound, because this is where Puget Sound Chinook, Puget Sound steelhead, and yelloweye rockfish, canary rockfish and bocaccio of the Puget Sound/Georgia Basin DPScould be exposed to the effects of the action. The outer boundaries are roughly defined by results of a three dimensional model simulation of the Nisqually Reach conducted by the Washington Department of Ecology (Figure 3).

This area includes a portion of South Puget Sound east of Anderson Island, as far south as the Nisqually River Delta, and as far north as Cormorant Passage in the Carr/Nisqually Subbasin (Figure 4).

For Southern Resident killer whales, the action area is defined by the extent of indirect effects on the killer whales that includes a marine area from the southern Strait of Georgia to southern Puget Sound and the Strait of Juan de Fuca, as well as from the mouth of the Strait of Juan de Fuca to the southern west coast of Vancouver Island (extent of range overlap between the killer whales and affected Puget Sound Chinook). This is the area in which the Southern Residents might encounter and consume Chinook that are exposed to the effluent from the proposed action.

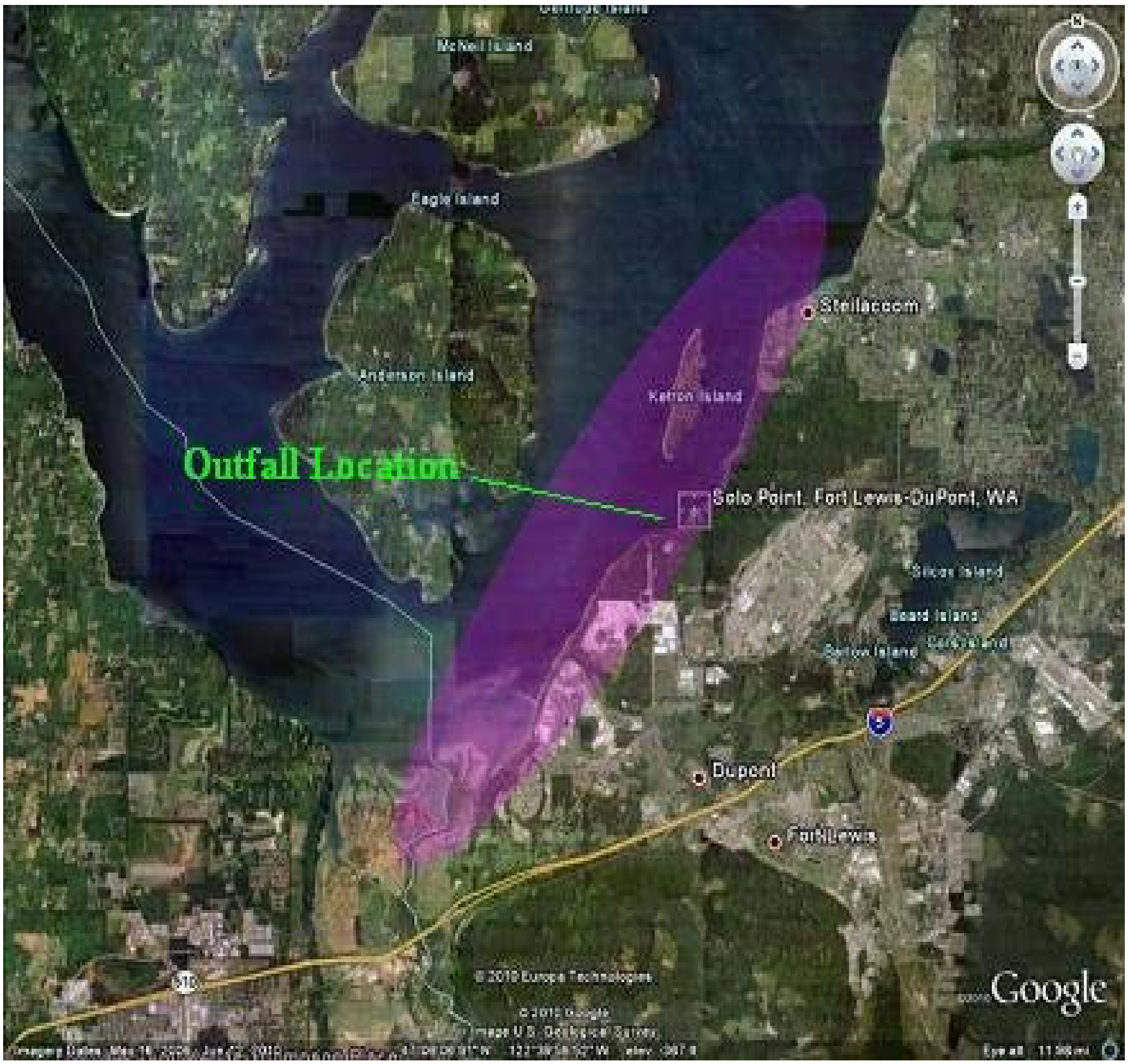


Figure 3: Action Area

The action area also includes designated critical habitat for both Puget Sound Chinook salmon and Southern Resident killer whales. Designated critical habitat for Puget Sound Chinook salmon extends from the line of extreme high tide out to the maximum depth of the photic zone, no greater than 30 meters relative to the MLLW (70 FR 52630). Southern Resident killer whale critical habitat is described below in the Status of the Species section.

This near shore element of the action area has also been identified as EFH for Chinook salmon, Puget Sound pink (*O. gorbuscha*), and coho salmon (*O. kisutch*), 16 groundfish species, and four coastal pelagic species.

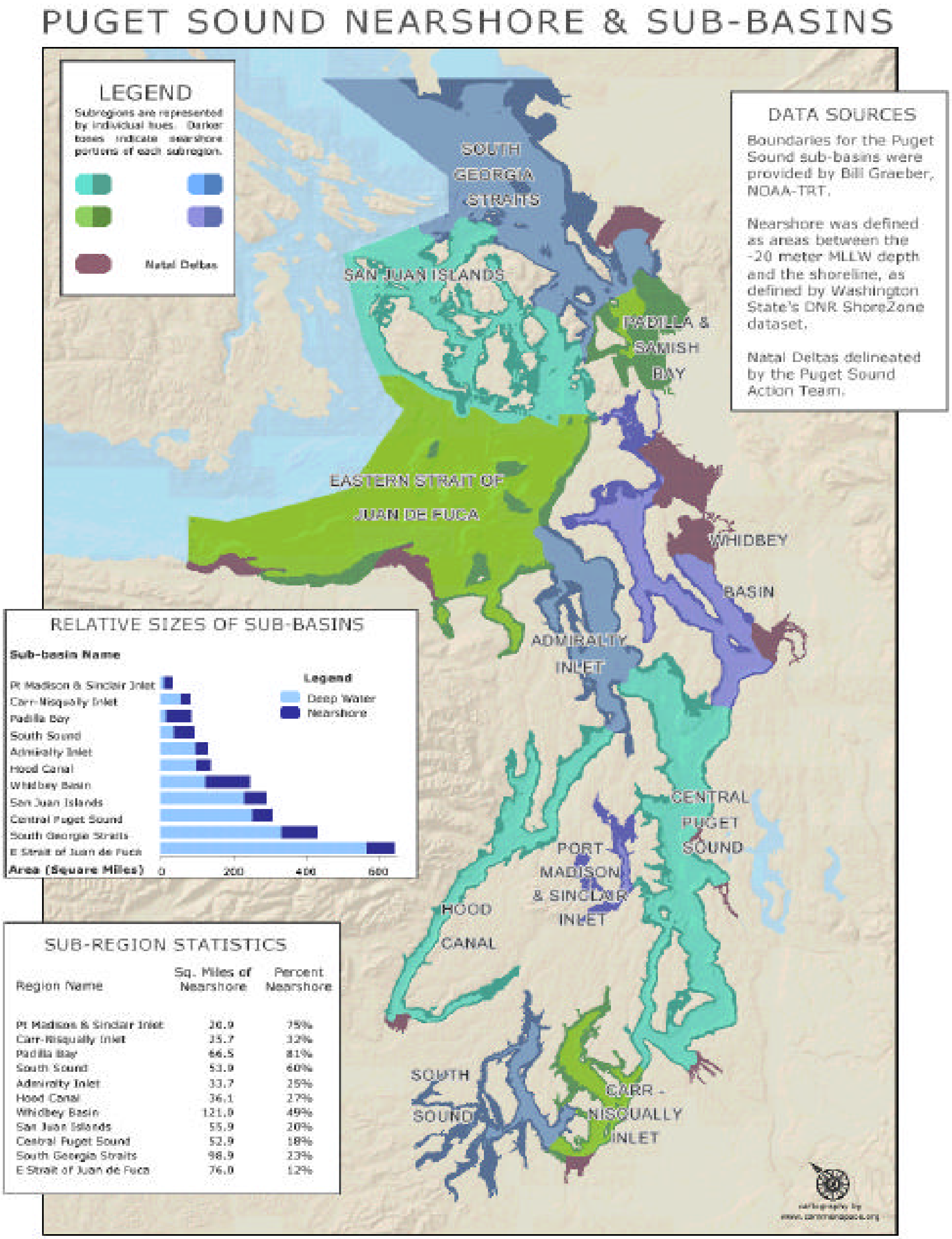


Figure 4: Puget Sound Subbasins

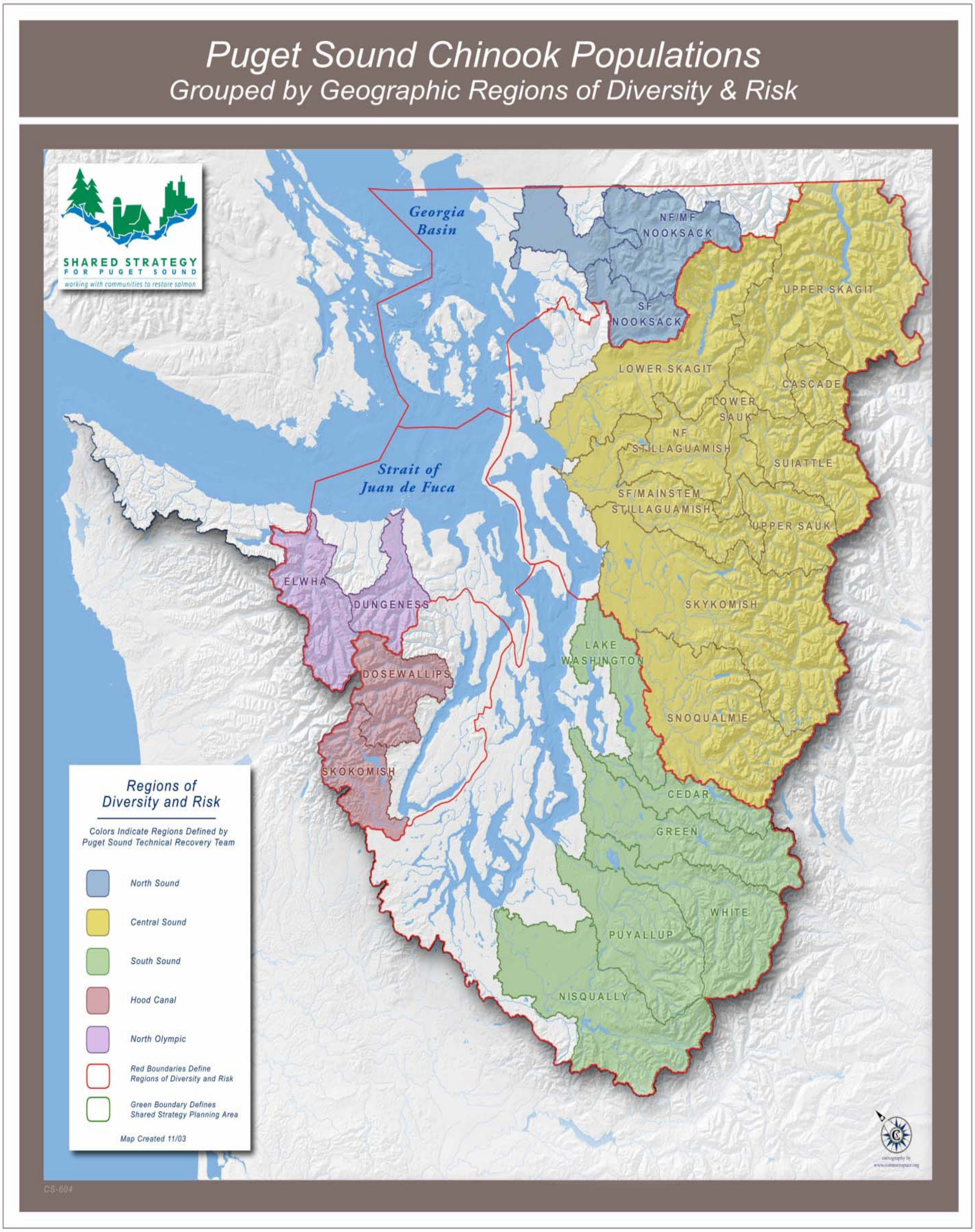


Figure 5. Puget Sound Chinook Salmon Populations

2. ENDANGERED SPECIES ACT: BIOLOGICAL OPINION AND INCIDENTAL TAKE STATEMENT

The ESA establishes a national program for conserving threatened and endangered species of fish, wildlife, plants, and the habitat on which they depend. Section 7(a) (2) of the ESA requires Federal agencies to consult with the United States Fish and Wildlife Service, NMFS, or both, to ensure that their actions are not likely to jeopardize the continued existence of endangered or threatened species or adversely modify or destroy their designated critical habitat. Section 7(b) (3) requires that at the conclusion of consultation, the Service provide an opinion stating how the agencies' actions will affect listed species or their critical habitat. If incidental take is expected, Section 7(b) (4) requires the provision of an incidental take statement (ITS) specifying the impact of any incidental taking, and including reasonable and prudent measures to minimize such impacts.

2.1 Analytical Approach to the Biological Opinion

Section 7(a) (2) of the ESA requires Federal agencies, in consultation with NMFS, to insure that their actions are not likely to jeopardize the continued existence of endangered or threatened species, or adversely modify or destroy their designated critical habitat. "To jeopardize the continued existence of a listed species" means to engage in an action that would be expected, directly or indirectly, to reduce appreciably the likelihood of both the survival and recovery of a listed species in the wild by reducing the reproduction, numbers, or distribution of that species (50 CFR 402.02). The jeopardy analysis considers both survival and recovery of the species.

To complete the jeopardy analysis presented in this opinion, NMFS reviewed the range-wide status of each listed species of Pacific salmon and steelhead, yelloweye rockfish, canary rockfish, bocaccio, and Southern Resident killer whales likely to be adversely affected by the proposed action, the environmental baseline in the action area, the effects of the action as proposed, and cumulative effects (50 CFR 402.14(g)). The effects of the action were added to the environmental baseline, along with the cumulative effects, to assess whether the action could reasonably be expected to appreciably reduce the likelihood of both survival and recovery of the species in the wild by reducing its numbers, reproduction, or distribution.

The adverse modification analysis considers the impacts to the conservation value of the designated critical habitat. This biological opinion does not rely on the regulatory definition of 'destruction or adverse modification' of critical habitat at 50 C.F.R. 402.02. Instead, we have relied upon the statutory provisions of the ESA to complete the following analysis with respect to critical habitat.[1] The adverse modification of critical habitat analysis includes a review of the status of critical habitat range-wide, the role of the environmental baseline, and evaluation of the

[1] Memorandum from William T. Hogarth to Regional Administrators, Office of Protected Resources, NMFS (Application of the "Destruction or Adverse Modification" Standard Under Section 7(a)(2) of the Endangered Species Act) (November 7, 2005).

proposed actions effects on Primary Constituent Elements (PCEs) of critical habitat. The anticipated effects, together with the anticipated cumulative effects on PCEs, are added to the baseline to determine if these would reduce the value of designated or proposed, critical habitat for the conservation of the species.

2.2 Range-wide Status of the Species and Critical Habitat

This section presents information about the status of listed species and their designated critical habitats. It evaluates the status and trend of listed species, using attributes associated with their viability criteria (McElhany et al., 2000), including information about their recovery. These attributes are influenced by survival, behavior, and experiences throughout the entire life cycle, which are influenced by habitat and other environmental conditions.

Throughout Washington, salmonids, rockfish, their forage, and their habitats are generally affected by climate change. Changes in climate conditions can also affect species that depend on salmon for prey such as Southern Resident killer whales. Climate change has been well documented in the scientific literature (Intergovernmental Panel on Climate Change [IPCC], 2007; Independent Scientific Advisory Board [ISAB], 2007). Several studies have revealed that climate change has the potential to affect ecosystems in nearly all tributaries throughout the state (ISAB 2007; Battin et al. 2007). While the intensity of effects will vary by region (ISAB 2007), climate change is generally expected to alter aquatic habitat (water yield, peak flows, and stream temperature). Evidence includes increases in average air and ocean temperatures, widespread melting of snow and glaciers, and rising sea level. Observations consistent with a changing global climate have already been documented in changes of species ranges and in a wide array of environmental trends (ISAB, 2007; Hari et al. 2006; Rieman et al. 2007). In the northern hemisphere, durations of ice cover over lakes and rivers have decreased by almost 20 days since the mid-1800s. These changes in snow pack decrease ocean productivity in the marine environment (ISAB 2007; Scheurell & Williams 2005).

As climate change alters the structure and distribution of rainfall, snowpack, and glaciations, each factor will in turn alter riverine hydrographs. Given the increasing certainty that climate change is occurring and is accelerating (Battin et al. 2007), NMFS anticipates salmonid habitats will be affected. An assessment by (O'Neal 2002) of the potential impacts of climate warming on salmon and trout habitat for the Pacific Northwest suggests a substantial decline in the habitats suitable for cold water fishes. Salmon habitat may be severely affected, in part because these fishes can only occupy areas below barriers and are thus restricted to lower, warmer elevations within the region. Projected salmon habitat loss in Washington will be about 22 percent by 2090, which does not consider the associated impact of changing hydrology. Karl et al., (2009) predict approximately one-third of the current salmon habitat in the Pacific Northwest will no longer be suitable by the end of this century due to climate change. The extent to which anadromous fish encounter serious adverse effects from changing hydrology will vary for each watershed.

In Washington State, most models project warmer air temperatures, increases in winter precipitation, and decreases in summer precipitation. Average temperatures in Washington State are likely to increase between 3.1 and 5.3 degrees Fahrenheit by 2040 (Casola et al. 2005). Warmer air temperatures will lead to more precipitation falling as rain rather than snow. As the snow pack diminishes, seasonal hydrology will shift to more frequent and severe early large storms, changing stream flow timing and increasing peak stream flows, which may limit salmon survival (Karl et al. 2009; NMFS 2008b). The largest driver of climate-induced decline in salmon populations in rivers is projected to be the impact of increased winter peak flows, which scour the streambed and destroy salmon eggs (Battin et al. 2007). Higher water temperatures and lower spawning flows, together with increased magnitude of winter peak flows are all likely to increase salmon mortality. Higher ambient air temperatures will likely cause water temperatures to rise (ISAB 2007). Salmon and steelhead require cold water for spawning and incubation. As climate change progresses and stream temperatures warm, thermal refugia will be essential to persistence of many salmonid populations. Thermal refugia are important for providing salmon and steelhead with patches of suitable habitat while allowing them to undertake migrations through or to make foraging forays into areas with greater than optimal temperatures. To avoid waters above summer maximum temperatures, juvenile rearing may be increasingly found only in the confluence of colder tributaries or other areas of cold water refugia (EPA, 2003).

Climate change may also adversely affect the Puget Sound/Georgia Basin ESA-listed rockfish and other obligate marine species. Important changes have occurred in the Puget Sound region in the past century and the next several decades will likely see even greater changes (Mote et al. 2005 as reported in Drake et al. 2010). Since the late 1800s, Pacific Northwest temperatures rose faster than the global average, and Puget Sound waters have warmed substantially since the early 1970s (Ruckelshaus and McClure 2007 as reported in Drake et al. 2010). Given the general importance of climate to rockfish recruitment, it is likely that climate strongly influences the dynamics of the ESA-listed rockfish population productivity and therefore their overall population viability (Drake et al. 2010). Recent declines in marine fish populations in greater Puget Sound, including rockfish, may reflect recent climatic shifts; however, it is not known whether these climatic shifts represent long-term changes or short-term fluctuations that may reverse in the near future (Drake et al. 2010). Potential long-term threats to ESA-listed rockfish species as a result of climate change, coupled with other threats such as by-catch by other fisheries, habitat loss, pollutants, and low dissolved oxygen (Drake et al. 2010) could further affect the survival and reproductive success of rockfish and their prey sources in the Puget Sound/Georgia Basin DPSs.

Ocean acidification may also affect ESA-listed rockfish and other rockfish and marine species in the Puget Sound/Georgia Basin. Ocean acidification is occurring globally and locally in response to increased carbon dioxide concentrations in the Earth's atmosphere. Carbon dioxide readily dissolves in marine waters and is readily converted to carbonic acid. This chemical reaction, driven by the saturation of marine waters with carbon dioxide, leads to a reduction in the pH of marine waters, or 'acidification'. For marine animals, including some fish, accumulation of CO_2 in the body may also result in changes in the organism's morphology, metabolic state, physical activity, and reproduction (Symposium on the Ocean in a High CO_2 World 2008). Perhaps more importantly, acidification of marine waters can affect, in particular, the viability of calcifying organisms such as bivalve shellfish and crustaceans in their pelagic larval stages by preventing their calcification and benthic settlement. The larval stages of these invertebrates during their planktonic phases provide forage for a myriad of fish species, particularly during their early life history stages. The impact of the loss of these forage resources for juvenile salmonids and rockfish remains to be evaluated. Any changes to these fish stocks as a result of climate change could indirectly affect the marine mammals that forage upon them, including southern resident killer whales and other listed and non-listed piscivorous marine mammals.

Climate change, and its downstream effects such as reduced winter snowpack, elevated water temperature, and ocean acidification, is expected to make recovery targets for salmon and rockfish populations more difficult to achieve. Habitat actions can potentially counter some of the adverse impacts of climate change, though the full measure of the response of these actions on population viabilities in the face of climate change is difficult to predict. Notwithstanding, habitat enhancement examples particularly relevant to salmonids include: restoring connections to historical floodplains and freshwater and estuarine habitats to provide fish refugia and areas to store excess floodwaters; protecting and restoring riparian vegetation to ameliorate stream temperature increases; and purchasing or applying easements to lands that provide important cold water or refuge habitat (ISAB 2007; Battin et al. 2007). Examples relevant to rockfish include restoring kelp bed habitats and developing conservation areas where harvest is restricted.

2.2.1 Status of the Species

Puget Sound Chinook Salmon Evolutionarily Significant Unit

The NMFS has compiled the following status summary based primarily on information from the status reviews of Myers et al. 1998, Good et al. 2005, NMFS 2010a, Ford et al. 2010), and the recovery plan (Shared Strategy 2007). This ESU was identified and originally assessed as part of the Chinook salmon coastwide status review in 1998 (Myers et al. 1998). It was reassessed in 2005 (Good et al. 2005) and again in 2010 (Ford et al. 2010). The ESU was listed as a threatened species on March 24, 1999 and the threatened status was reaffirmed on June 28, 2005, and again in 2011 following the publication of the most recent status review by Ford et al. (Table 1).

Good et al. (2005) as reiterated in Ford et al. (2010) summarized that the natural spawning escapement for Puget Sound Chinook salmon populations were slightly improved relative to those at the time of the previous status review of Puget Sound Chinook salmon conducted with data through 1997 (Myers et al. 1998). The overall trends in natural spawning escapements for Puget Sound Chinook salmon populations estimated in 2005 remained similar to that presented in the previous status review (data through 1997), with some populations doing marginally better and others worse. The 2010 review of Ford et al. (2010) found similar results.

Spatial Structure and Diversity. The Puget Sound ESU includes all naturally spawned populations of Chinook salmon from rivers and streams flowing into Puget Sound including the Strait of Juan De Fuca from the Elwha River, eastward, including rivers and streams flowing into Hood Canal, South Sound, North Sound and the Strait of Georgia in Washington, as well as twenty-six artificial propagation Programs (Figure 5).

This ESU is composed of 31 historically quasi-independent populations, 22 of which are believed to be extant (Puget Sound TRT 2001; Ruckelshaus et al. 2006). The populations presumed extinct are mostly early returning fish that for the most part occur in mid- to Southern Puget Sound or Hood Canal and the Strait of Juan de Fuca. The ESU populations with the greatest estimated fractions of hatchery fish tend to be in Central and South Puget Sound, Hood Canal, and the Strait of Juan de Fuca.

Eight of 26 existing artificial propagation programs are directed at conserving Puget Sound Chinook salmon. The remaining programs considered to be part of the ESU are operated primarily for fisheries harvest augmentation purposes (some of which also function as research programs) using transplants of within-ESU-origin Chinook salmon as broodstock.

The NMFS determined that these artificially propagated stocks are no more divergent relative to the local natural population(s) than what would be expected between closely related natural populations within the ESU (NMFS 2005).

Good et al. (2005) found that the spatial distribution of Chinook salmon populations with a strong component of natural-origin spawners in this ESU had not changed since the last status assessment by Myers et al., (1998). Populations containing significant numbers of natural-origin spawners, whose status can be reliably estimated, occur in the Skagit, South Fork Stillaguamish, and Snohomish river basins. The remaining populations in mid- and South Puget Sound, Hood Canal, and the Strait of Juan de Fuca have significant, but non-quantifiable, fractions of hatchery origin spawners, so their contribution to spatial structure in the ESU is not possible to estimate. Good et al., (2005) also found that diversity had not changed since the last status assessment.

Abundance and Productivity. Overall abundance of Chinook salmon in this ESU has declined substantially from historical levels of approximately 690,000 spawners in the early 1900s. Several populations are small enough that genetic and demographic risks are high. In its 1998 status review, NMFS noted that the average run size (hatchery plus natural) at that time was approximately 240,000 fish, with natural spawning escapement averaging 25,000 fish (Myers et al. 1998), natural spawning escapement has increased in all life history types (spring, summer and fall) to an annual average of approximately 45,000 fish.

Since listing, the geometric mean (1999–2009) of natural spawners ranges from 150 (Mid-Hood Canal population) to just over 10,000 fish (Upper Skagit River population). Thirteen of the 22 populations contain natural spawners numbering over 1,000 fish (median recent natural escapement = 1,254 fish) and all but two are well below the recovery targets. On average, the abundance (geometric mean of natural spawners) is at 58 percent of the recovery targets (NMFS 2010a).

Estimates of the fraction of hatchery origin, natural spawners are currently limited. Data are available for 19 of the 22 populations in the ESU, but its quality varies greatly and only covers the most recent 5-10 year period. Based on this information, the six Skagit populations have very little hatchery contribution to natural spawning. The Cedar, Duwamish-Green, White, Puyallup, Stillaguamish and Snohomish populations have moderate proportions of naturally spawning hatchery fish. The Nooksack, Sammamish, Nisqually, and Dungeness have substantial numbers of naturally spawning hatchery fish. More comprehensive information will become available over the next several years as management agencies have increased marking and monitoring programs to track hatchery fish (NMFS 2010a).

Nineteen populations exhibit a stable or increasing growth rate in escapement with the White River population showing a significant increasing trend. Growth rates are declining for the South Fork Stillaguamish, Sammamish, and Puyallup populations. No clear patterns are evident with regard to trends in abundance or growth rate among the five major regions of Puget Sound (NMFS 2010a).

All Puget Sound Chinook populations are well below the Puget Sound TRT planning range for recovery escapement levels (Ford et al. 2010). Most populations are also consistently below the spawner-recruit levels identified by the TRT as consistent with recovery. Across the ESU, most populations have declined in abundance somewhat since the last status review in 2005, and trends since 1995 are mostly flat. Several of the risk factors identified by Good et al. (2005) remain, including high fractions of hatchery fish in many populations and widespread loss and degradation of habitat. Many of the habitat and hatchery actions identified in the Puget Sound Chinook recovery plan are expected to take years or decades to be implemented and to produce significant improvements in natural population attributes, and these trends are consistent with these expectations. Overall, information presented by Ford et al. (2010) on abundance, productivity, spatial structure and diversity since the 2005 review does not indicate a change in the biological risk category since the 2005 BRT status review.

Puget Sound Steelhead Distinct Population Segment

The NMFS compiled the following summary based on information from the most recent Puget Sound steelhead status review (Ford et al. 2010), unless otherwise noted.

Spatial Structure and Diversity. The Puget Sound steelhead DPS includes all naturally spawned anadromous winter-run and summer-run steelhead populations, from streams in the river basins of the Strait of Juan de Fuca, Puget Sound, and Hood Canal, Washington, bounded to the west by the Elwha River (inclusive) and to the north by the Nooksack River and Dakota Creek (inclusive), as well as the Green River natural and Hamma Hamma winter-run steelhead hatchery stocks (72 FR 26722; May 11, 2007). The majority of hatchery stocks are not considered part of this DPS because they are more than moderately diverged from the local native populations (NMFS 2005). Resident steelhead occur within the range of Puget Sound steelhead but are not part of the DPS due to marked differences in physical, physiological, ecological, and behavioral characteristics (71 FR 15666; March 29, 2006).

Although the Puget Sound steelhead DPS includes more than 50 stocks of summer- and winter-run fish, the DPS is composed primarily of winter-run populations (Hard et al. 2007). Populations of summer run steelhead occur throughout the Puget Sound DPS but are concentrated in northern Puget Sound area, are generally small, and are characterized as isolated populations adapted to streams with distinct attributes. For the one summer-run population that has associated natural escapement and run size data, the trend in abundance was predominantly negative (Hard et al. 2007). There is also concern that some historical accounts discuss significant early runs of wild fish, but that these early wild spawners have apparently disappeared from several systems. Spatial structure of steelhead in the DPS poses moderate risk to its viability. The DPS is likely to be at elevated risk due to reduced complexity of spatial structure of its steelhead populations, and consequently, diminishing connectivity among them (Hard et al. 2007).

Abundance and Productivity. The Biological Review Team (BRT) (Hard 2007) concluded that low and declining abundance and productivity were substantial risk factors for Puget Sound steelhead. Marked declines in natural run-size are evident in all areas of the DPS, a pattern that reflects widespread reduced productivity of natural steelhead. Estimates of lambda were less than 1.0 for nearly all populations in the DPS, indicating declining population growth. Most recently, Ford et al. 2010 found only three of the 16 populations evaluated exhibit point estimates of growth rate that are positive (East Hood Canal, Skokomish River, and West Hood Canal), and only one of these is significantly greater than one (positive population growth): West Hood Canal. These four populations are all small. The highest growth rates from 1995-2009 occur in East Hood Canal, the Skokomish River, and the Samish and Skagit rivers; the lowest rates occur in the Elwha and Dungeness rivers, Lake Washington, and the Stillaguamish, Nisqually, and Puyallup rivers. Trends could not be calculated for south Puget Sound tributaries. Winter-run steelhead in the Skagit and Snohomish rivers support the two largest populations in the DPS, averaging approximately 5,000 (Skagit) and 3,000 (Snohomish) total adult spawners annually (Ford et al. 2010).

Over the most recent five years (2005-2009), Puget Sound winter-run steelhead abundance has been low over much of the DPS, with a geometric mean less than 250 fish annually for all but eight populations of the 15 evaluated. Four of these are in northern Puget Sound (Samish, Skagit, Snohomish and Stillaguamish rivers), three are in southern Puget Sound (Nisqually, Puyallup, and White rivers), and one is on the Olympic Peninsula (Skokomish River). Only three populations have a geometric mean greater than 500 fish—Green, Skagit and Samish rivers—and two of these are in northern Puget Sound. The Elwha River, Lake Washington, and South Sound tributaries populations all have very low recent mean abundances (fewer than 15 fish).

Collectively, these data indicate relatively low abundance (4 of 15 populations with fewer than 500 spawners annually) and declining trends (6 of 16 populations) in natural escapement of winter-run steelhead throughout Puget Sound, particularly in southern Puget Sound and on the Olympic Peninsula (Ford et al. 2010).

Continued releases of out-of-ESU hatchery fish from Skamania-derived summer-run and Chambers Creek-derived winter-run stocks are a major diversity concern. Hatchery fish in this DPS are widespread, and spawn naturally throughout the region. Hard et al. (2007) found that the proportion of spawning escapement comprised of hatchery fish ranged from less than 1 percent (Nisqually River) to 51 percent (Morse Creek). In general, hatchery proportions are higher in Hood Canal and the Strait of Juan de Fuca than in Puget Sound proper. Most of the hatchery fish in this region originated from stocks indigenous to the DPS, but are generally not native to local river basins. Summer steelhead stocks within this DPS are all small, occupy limited habitat, and most are subject to introgression by hatchery fish (Hard et al. 2007).

Declines in natural abundance for most populations, coupled with large numbers of anthropogenic barriers such as impassable culverts reduce opportunities for movement and migration between steelhead aggregations in different watersheds (Hard et al. 2007). The reduction in escapement of natural steelhead to the centrally located Lake Washington watershed in recent years is also of concern, especially due to weakening trends in abundance for neighboring populations.

For all but a few demographically independent populations of steelhead in Puget Sound, estimates of mean population growth rates obtained from observed spawner or redd counts are declining—typically 3 to 10% annually—and extinction risk within 100 years for most populations in the DPS is estimated to be moderate to high, especially for populations in South Sound and Olympic Peninsula. Collectively, these analyses indicate that steelhead in the Puget Sound DPS remain at risk of extinction throughout all or a significant portion of their range in the foreseeable future, but are not currently in danger of imminent extinction (Ford et al. 2010).

In summary, the current status of the listed Puget Sound steelhead DPS as summarized by Ford et al. (2010) has not changed substantially since the 2007 listing. Most populations within the DPS are showing continued downward trends in estimated abundance, a few sharply so.

Puget Sound/Georgia Basin Distinct Population Segments of Yelloweye Rockfish, Canary Rockfish and Bocaccio

The Puget Sound/Georgia Basin DPSs include all yelloweye rockfish, canary rockfish and bocaccio found in waters of the Puget Sound, the Strait of Georgia, and the Strait of Juan de Fuca east of Victoria Sill. Puget Sound is the second-largest estuary in the United States, located in northwest Washington State, covering an area of about 2,600 square kilometers (1,020 square miles), including 4,000 kilometers (2,500 miles) of shoreline and is home to a rapidly-expanding human population. Puget Sound is part of a larger inland waterway, the Georgia Basin, situated between Southern Vancouver Island, British Columbia, Canada and the mainland coasts of Washington State. Puget Sound can be subdivided into five interconnected basins separated by shallow sills: (1) the San Juan/Strait of Juan de Fuca region; (2) Main Basin; (3) Whidbey Basin; (4) South Puget Sound; and (5) Hood Canal. The NMFS uses the term ‘‘Puget Sound Proper’’ to refer to all of these basins except the San Juan/Strait of Juan de Fuca region. These basins have unique temperature regimes, water residence times and circulation patterns, biological conditions, depth profiles and contours, species compositions, and nearshore and benthic habitats (Ebbesmeyer 1984; Burns 1985; Rice 2007).

Life Histories. The life-histories of the yelloweye rockfish, canary rockfish and bocaccio include a larval and pelagic juvenile stage followed by a nearshore juvenile stage and sub-adult and adult stage. Much of the life-history and biological requirements for these three species is similar, with differences noted below.

Larval and Pelagic Juvenile Stage. Rockfish fertilize their eggs internally and the young are extruded as larvae. As larvae, rockfish generally occupy the upper portion of the water column and are often near the surface (Love et al. 2002) but can be found throughout the water column (Weis 2004). Larvae can make small local movements to pursue food immediately after birth (Tagal et al. 2002), but are nonetheless passively distributed with prevailing currents (NMFS 2003). Larvae are often observed under free-floating algae, seagrass and detached kelp (Shaffer et al. 1995; Love et al. 2002). Unique oceanographic conditions within Puget Sound Proper likely result in most larvae staying within the region where they are born rather than being dispersed to adjacent regions (Drake et al. 2010).

Though there is a dearth of studies that have sampled for rockfish larvae presence in Puget Sound outside of the spring time, larval rockfish do occur throughout the year along the Pacific Coast and very likely occupy the action area throughout the year (Waldron 1972; Westrheim & Harling 1975; Wylie Echerverria 1987; Moser & Boehert 1991; Love et al. 2002; Weis 2004). Each species produces from several thousand to over a million eggs within one birth event (Love et al. 2002). The term 'cohort' is typically used when referring to larvae released within one birth event from one mother.

Larval rockfish are extremely fragile and mortality rates range from approximately 21 percent to 50 percent per day immediately after birth (Weis 2004) and rises to 70 percent 7- 12 days after birth (Canino & Francis 1989). Their small size, relative inability to store food within their gut, and slow swimming speeds likely contribute to this high mortality rate by making them vulnerable to predators and starvation. Predators of larval rockfish include herring, surf smelt, salmon, and other fish.

Nearshore Juvenile Stage. When bocaccio and canary rockfish reach sizes of 3 to 9 centimeters or 3 to 6 months old, they settle onto shallow nearshore waters in rocky or cobble substrates with or without kelp (Love et al. 1991; Love et al. 2002). These habitats likely feature a beneficial mix of warmer temperatures, food, and refuge from predators (Love et al. 1991). Areas with floating and submerged kelp species support the highest densities of most juvenile rockfish (Carr 1983; Halderson & Richards 1987; Matthews 1989; Hayden-Spear 2006). Unlike bocaccio and canary rockfish, juvenile yelloweye rockfish do not typically occupy intertidal waters (Love et al. 1991; Studebaker et al. 2009), but settle in 100 to 130 feet of water near the upper depth range of adults (Yamanaka & Lacko 2001).

Sub-Adult and Adult. Sub-adult and adult yelloweye rockfish, canary rockfish and bocaccio typically utilize habitats with moderate to extreme steepness, complex bathymetry and rock and boulder-cobble complexes (Love et al. 2002). Within Puget Sound Proper, each species has been documented in areas of high relief rocky and non-rocky substrates such as sand, mud and other unconsolidated sediments (Washington 1978; Miller & Borton 1980; WDFW unpublished data). Yelloweye rockfish remain near the bottom and have small home-ranges, while some canary rockfish and bocaccio have larger home ranges, move long distances, and spend time suspended in the water column (Love et al. 2002). Adults of each species are most commonly found deeper than 120 feet (Love et al. 2002; Orr et al. 2000).

In southeast Alaska, adult yelloweye and canary rockfish were observed at mean depths of 150 feet and 173 feet, and minimum depths of 69 and 121 feet, respectively (Johnson et al., 2003). Yelloweye rockfish are one of the longest lived of the rockfishes, reaching more than 100 years of age, and reach 50 percent maturity at sizes around 40 to 50 centimeters and ages of 15 to 20 years (Rosenthal et al. 1982, Yamanaka & Kronlund 1997). Maximum age of canary rockfish is at least 84 years (Love et al. 2002), although 60 to 75 years is more common (Caillet et al. 2000). They reach 50 percent maturity at sizes around 40 centimeters and ages of 7 to 9. The maximum age of bocaccio is unknown, but may exceed 50 years, and they are first reproductively mature near age 6 (Love et al. 2002).

The timing of larval release for each species varies throughout the geographic range. In Puget Sound, there is some evidence that larvae are extruded in early spring to late summer for yelloweye rockfish (Washington et al. 1978). In British Columbia, parturition (larval birth) peaks in February for canary rockfish (Westrheim & Harling 1975). Along the coast of Washington state, female bocaccio release larvae between January and April (Love et al. 2002). Each species produces from several thousand to over a million eggs (Love et al. 2002).

Viability Criteria. In the following section, NMFS summarizes the condition of the yelloweye rockfish, canary rockfish and bocaccio DPSs at the Puget Sound level according to the following demographic risk criteria: abundance and productivity, spatial structure and connectivity, and diversity. These viability criteria are outlined in McElhaney et al. (2000), and reflect concepts that are well founded in conservation biology and are generally applicable to a wide variety of species. These criteria describe demographic risks that individually and collectively provide strong indicators of extinction risk (Drake et al. 2010).

Abundance & Productivity. The abundance of individuals in a population is important in assessing two aspects of extinction risk. First, population size can be an indicator of whether the population can sustain itself in the face of environmental fluctuations and small-population stochasticity, even if it currently may be stable or increasing. Second, abundance in a declining population is an indicator of the time expected until the population reaches critically low numbers (Drake et al. 2010). Small rockfish populations are subject to additional risks that include: (1) environmental variation such as altered temperature regimes and circulation patterns that could disrupt food-webs, larval dispersal or juvenile rearing; (2) genetic processes, such as the accumulation of negative mutations; (3) demographic stochasticity, such as imbalanced gender ratios; (4) ecological feedback, such as other fish species occupying the niche left by the depleted population which hinders recovery; and (5) catastrophes, such as oil spills, which disrupt benthic environments or larval/juvenile rearing habitats and food sources (McElhaney et al. 2000). An additional risk from low abundance is the depensatory processes (termed "Allee" effects) that occurs when mates cannot find one another (Courchamp et al. 2008).

There is no single reliable historic or contemporary population estimate for yelloweye rockfish, canary rockfish or bocaccio within the DPSs (Drake et al. 2010). Despite this limitation, there is clear evidence each species' abundance has declined dramatically (Drake et al. 2010). The total rockfish population in the Puget Sound region is estimated to have declined around three percent per year for the past several decades, which corresponds to an approximate 70 percent decline from the 1965 to 2007 time period (Drake et al. 2010). Catches of each species have declined as a proportion of the overall rockfish catch (Palsson et al. 2009; Drake et al. 2010).

Fishery-independent estimates of population abundance come from spatially and temporally limited research trawls, drop camera surveys and underwater remotely operated vehicle surveys conducted by WDFW. Using these methods, WDFW has estimated that 50,655 yelloweye rockfish, 20,449 canary rockfish and 4487 bocaccio are within the Puget Sound region (NMFS, 2010). Most of the fish observed by the above sampling methods that inform population estimates were in the San Juan portion of the DPSs. These population estimates have generally large variances (or standard errors), and thus there remains uncertainty regarding the total abundance and distribution of ESA-listed rockfish within the Puget Sound/Georgia Basin DPSs of each species.

Productivity is the measurement of a population's growth rate through all or a portion of its life-cycle. Life-history traits of yelloweye rockfish, canary rockfish and bocaccio suggest generally low levels of inherent productivity because they are long-lived and mature slowly, with sporadic episodes of successful reproduction (Tolimieri & Levin 2005; Drake et al. 2010). Historic over fishing can have dramatic impacts on the size or age structure of the population, with effects that can influence ongoing productivity. As the size and age of females' decline, there are negative impacts on overall reproductive success. These impacts, termed maternal effects, are evident in a number of traits. Larger and older females of various rockfish species have a higher weight-specific fecundity (number of larvae per unit of female weight) (Bobko & Berkeley 2004; Sogard et al. 2008; Boehlert et al. 1982).

A consistent maternal effect in rockfishes relates to the timing of parturition. The timing of larval release can be crucial in terms of matching favorable oceanographic conditions for larvae because most are released on only 1 day each year, with a few exceptions in southern coastal populations and yelloweye in Puget Sound (Washington et al. 1978). Larger or older females release larvae earlier in the season compared to smaller or younger females in several studies of rockfish species (Nichol & Pikitch 1994; Sogard et al. 2008). Larger or older females provide more nutrients to larvae by developing a larger oil globule released at parturition, which provides energy to the developing larvae (Berkeley et al. 2004; Fisher et al. 2007), and in black rockfish enhances early growth rates (Berkeley et al. 2004). An additional maternal effect in black rockfish indicates that older females are more successful in completing recruitment of progeny from primary oocyte to fully developed larva (Bobko & Berkeley 2004).

Contaminants such as polychlorinated biphenyls (PCBs), chlorinated pesticides, and polybrominated diphenyl ethers (PBDEs) appear in rockfish collected in urban areas (Palsson et al. 2009). While the highest levels of contamination occur in urban areas, toxins can be found in the tissues of fish throughout the region (Puget Sound Action Team 2007). Although few studies have investigated the effects of toxins on rockfish ecology or physiology, other fish in the Puget Sound region that have been studied do show a substantial impact, including reproductive dysfunction of some sole species (Landahl et al. 1997). Reproductive function of rockfish is also likely affected by contaminants (Palsson et al. 2009).

Spatial Structure and Connectivity. Spatial structure consists of a population's geographical distribution and the processes that generate that distribution (McElhaney et al. 2000). A population's spatial structure depends on habitat quality, spatial configuration, and dynamics as well as dispersal characteristics of individuals within the population (McElhaney et al. 2000).

The apparent steep reduction of ESA-listed rockfish in Puget Sound proper leads to concerns about the viability of these populations (Drake et al. 2010). Yelloweye rockfish spatial structure and connectivity is likely threatened by the apparently severe reduction of fish within all or portions of Hood Canal and the South Puget Sound, combined with their small home-ranges as adults. Similarly, several historically large aggradations of canary rockfish in Puget Sound have been depleted, including an area of historic distribution in South Puget Sound (Drake et al. 2010). Bocaccio may have been historically spatially limited to several regions within the Puget

Sound. They were apparently historically most abundant in the Central and South Puget Sound (Drake et al. 2010).

For canary rockfish and bocaccio, positive signs for spatial structure and connectivity come from the propensity of some adults and pelagic juveniles to migrate long distances, which could reestablish aggregations of fish in formerly occupied habitat (Drake et al. 2010).

Diversity. Characteristics of diversity for rockfish include fecundity, timing of the release of larvae and their condition, morphology, age at reproductive maturity and physiology and molecular genetic characteristics. In spatially and temporally varying environments, there are three general reasons why diversity is important for species and population viability: (1) diversity allows a species to use a wider array of environments; (2) it protects a species against short-term spatial and temporal changes in the environment; and (3) genetic diversity provides the raw material for surviving long-term environmental changes. Though there are no genetic data within the DPSs of ESA-listed rockfish, the unique oceanographic features and relative isolation of some of its regions may have led to unique adaptations, such as timing of larval release (Drake et al. 2010).

ESA-listed rockfish size (and age) distribution have been truncated. Recreationally caught yelloweye rockfish, canary rockfish and bocaccio in the 1970's spanned a broad range of sizes. By the 2000's, there was evidence of fewer older fish (Drake et al. 2010). For each species, the reproductive burden may be shifted to younger and smaller fish. This shift could alter the timing and condition of larval release, which may be mismatched with habitat conditions within the Puget Sound, potentially reducing the viability of offspring (Drake et al. 2010).

Southern Resident Killer Whales

The Southern Resident killer whale Distinct Population Segment (DPS), composed of J, K and L pods, was listed as endangered under the ESA on November 18, 2005 (70 FR 69903). Southern Residents are designated as "depleted" and "strategic" under the Marine Mammal Protection Act (MMPA) (68 FR 103, May 29, 2003).

This section summarizes the status of the Southern Residents throughout their range. The final recovery plan for Southern Residents was issued in January 2008 (NMFS 2008a). This section summarizes information taken largely from the recovery plan and recent five-year status review (NMFS 2011a) as well as new data that became available more recently. For more detailed information about this population, please refer to the Final Recovery Plan for Southern Resident Killer Whales, which can be found on the internet at www.nwr.noaa.gov.

Abundance, Productivity and Trends. Southern Resident killer whales are a long-lived species, with late onset of sexual maturity (review in NMFS 2008a). Females produce a low number of surviving calves over the course of their reproductive life span (Bain 1990; Olesiuk et al. 1990). Southern Resident females appear to have reduced fecundity relative to Northern Residents; the average inter-birth interval for reproductive Southern Resident females is 6.1 years, which is longer than that of Northern Resident killer whales (Olesiuk et al. 2005). Mothers and offspring maintain highly stable social bonds throughout their lives, which is the basis for the matrilineal social structure in the Southern Resident population (Baird 2000; Bigg et al. 1990; Ford et al.

2000). Groups of related matrilines form pods. Three pods – J, K, and L – make up the Southern Resident community. Clans are composed of pods with similar vocal dialects and all three pods of the Southern Residents are part of J clan.

The historical abundance of Southern Resident killer whales is estimated from 140 to an unknown upper bound. The minimum historical estimate (~140) included whales killed or removed for public display in the 1960s and 1970s added to the remaining population at the time the captures ended. Several lines of evidence (i.e., known kills and removals [Olesiuk et al. 1990], salmon declines [Krahn et al. 2002] and genetics [Krahn et al. 2002; Ford et al. 2011a]) all indicate that the population used to be a lot larger than it is now, but there is currently no reliable estimate of the upper bound of the historical population size. When faced with developing a population viability analysis for this population, NMFS' biological review team found it reasonable to assume an upper bound of as high as 400 whales to estimate carrying capacity (Krahn et al. 2004).

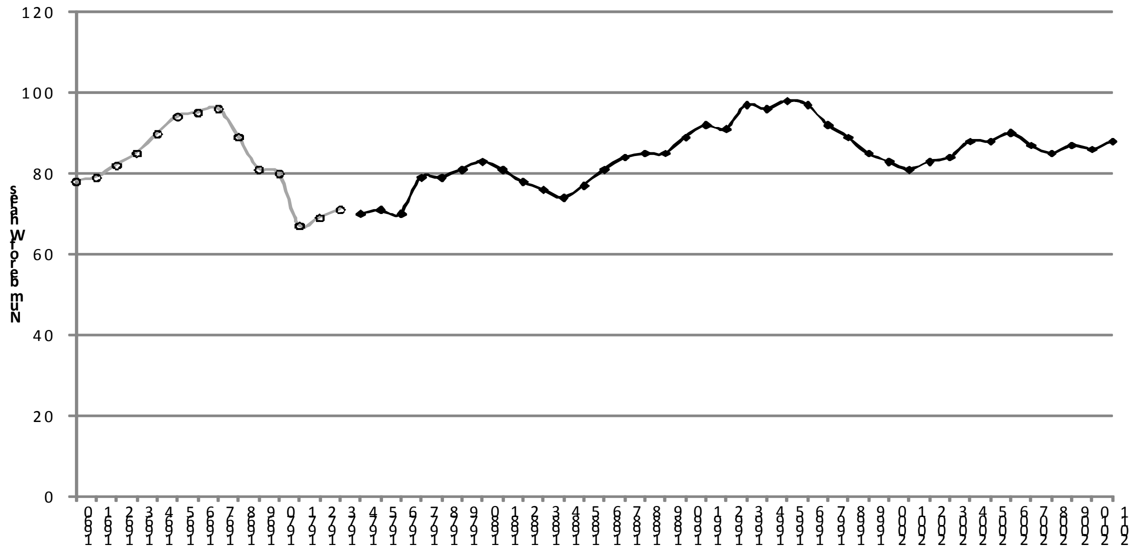
At present, the Southern Resident population has declined to essentially the same size that was estimated during the early 1960s, when it was considered as likely depleted (Olesiuk et al. 1990) (Figure 7). Since censuses began in 1974, J and K pods have steadily increased their sizes. However, the population suffered an almost 20 percent decline from 1996-2001 (from 97 whales in 1996 to 81 whales in 2001), largely driven by lower survival rates in L pod. Since then the overall population has increased slightly from 2002 to present (from 83 whales in 2002 to 88 whales as of August, 2011). Over the last 28 years (1983-2010), population growth has been variable, with an average annual population growth rate of 0.3 percent and standard deviation of ± 3.2 percent. Seasonal mortality rates among Southern and Northern Resident whales may be highest during the winter and early spring, based on the numbers of animals missing from pods returning to inland waters each spring. Olesiuk et al. (2005) identified high neonate mortality that occurred outside of the summer season. At least 12 newborn calves (9 in the southern community and 3 in the northern community) were seen outside the summer field season and disappeared by the next field season (NMFS 2008a). Additionally, stranding rates are higher in winter and spring for all killer whale forms in Washington and Oregon (Norman et al. 2004). Southern Resident strandings in coastal waters offshore include three separate events (1995 and 1996 off of Northern Vancouver Island and the Queen Charlotte Islands, and 2002 offshore of Long Beach, Washington State), but the causes of death are unknown (NMFS 2008a).

There is representation in all three pods, with 26 whales in J pod, 20 whales in K pod and 42 whales in L pod. There are currently 3 adult males and one nearly matured male in J pod, 3 adult males in K pod, and 9 adult males and nearly one matured male in L pod. The population is 37.5 percent juveniles, 33.0 percent reproductive females, 10.2 percent post-reproductive females and 19.3 percent adult males. This age distribution is similar to that of Northern Residents that are a stable and increasing population (Olesiuk et al. 2005). However, there are several demographic factors of the Southern Resident population that are cause for concern, namely the small number of breeding males (particularly in J and K pods), reduced fecundity, decreased sub-adult survivorship in L pod, and the total number of individuals in the population (review in NMFS 2008a). The current population abundance of 88 whales is small, at most half of its likely previous abundance (140 to an unknown upper bound that could be as high at 400 whales, as discussed above). The estimated effective size of the population (based on the number of breeders under ideal genetic conditions) is very small at 26 whales or roughly 1/3 of the current population size (Ford et al. 2011a). The small effective population size and the absence of gene flow from other populations may elevate the risk from inbreeding and other issues associated with genetic deterioration, as evident from documented breeding within pods (Ford et al. 2011a). As well, the small effective population size may contribute to the lower growth rate of the Southern Resident population in contrast to the Northern Resident population (Ford et al. 2011a; Ward et al. 2009).

Because of this population's small abundance, it is also susceptible to demographic stochasticity – randomness in the pattern of births and deaths among individuals in a population. Several other sources of stochasticity can affect small populations and contribute to variance in a population's growth and extinction risk. Other sources include environmental stochasticity, or fluctuations in the environment that drive fluctuations in birth and death rates, and demographic heterogeneity, or variation in birth or death rates of individuals because of differences in their individual fitness (including sexual determinations). In combination, these and other sources of random variation combine to amplify the probability of extinction, known as the extinction vortex (Gilpin and Soule 1986; Fagen and Holmes 2006; Melbourne and Hastings 2008). The larger the population size, the greater the buffer against stochastic events and genetic risks. A delisting criterion for the Southern Resident killer whale DPS is an average growth rate of 2.3percent for 28 years (NMFS 2008a). In light of the current average growth rate of 0.3percent, this recovery criterion reinforces the need to allow the population to grow quickly.

Population growth is also important because of the influence of demographic and individual heterogeneity on a population's long-term viability. Population-wide distribution of lifetime reproductive success can be highly variable, such that some individuals produce more offspring than others to subsequent generations, and male variance in reproductive success can be greater than that of females (i.e., Clutton-Brock 1988; Hochachka 2006). For long-lived vertebrates such as killer whales, some females in the population might contribute less than the number of offspring required to maintain a constant population size (n = 2), while others might produce more offspring. The smaller the population, the more weight an individual's reproductive success has on the population's growth or decline (i.e., Coulson et al. 2006). This further illustrates the risk of demographic stochasticity for a small population like Southern Resident killer whales – the smaller a population, the greater the chance that random variation will result in too few successful individuals to maintain the population.

Figure 7. Population size and trend of Southern Resident killer whales, 1960-2011. Data from 1960-1973 (open circles, gray line) are number projections from the matrix model of Olesiuk et al. (1990). Data from 1974-2011 (diamonds, black line) were obtained through photo-identification surveys of the three pods (J, K, and L) in this community and were provided by the Center for Whale Research (unpubl. data) and NMFS (2008a). Data for these years represent the number of whales present at the end of each calendar year, except for 2011 when data only extend to August.



Range and Distribution. Southern Residents are found throughout the coastal waters of Washington, Oregon, and Vancouver Island and are known to travel as far south as central California and as far north as the Queen Charlotte Islands, British Columbia (Figure 8). There is limited information on the distribution and habitat use of Southern Residents along the outer Pacific Coast.

Figure 8. Geographic Range (light shading) of the Southern Resident Killer Whale DPS. Reprinted from Wiles (2004).

Southern Residents are highly mobile and can travel up to 86 miles (160 km) in a single day (Erickson 1978; Baird 2000). To date, there is no evidence that Southern Residents travel further than 50 km offshore (Ford et al. 2005). Although the entire Southern Resident killer whale DPS has potential to occur in coastal waters at any time during the year, occurrence is more likely during November to May.

Southern Residents spend considerable time from late spring to early autumn in inland waterways of Washington State and British Columbia (Strait of Georgia, Strait of Juan de Fuca, and Puget Sound; Bigg 1982; Ford et al. 2000; Krahn et al. 2002, Table 2). Typically, J, K and L pods are increasingly present in May or June and spend considerable time in the core area of Georgia Basin and Puget Sound until at least September. During this time, pods (particularly K and L) make frequent trips from inland waters to the outer coasts of Washington and southern Vancouver Island, which typically last a few days (Ford et al. 2000).

Table 2. Average number of days spent by Southern Resident killer whales in inland and coastal waters[1] by month, 2003-2007 (Hanson and Emmons, 2010).

Months	Lpod		Jpod		Kpod	
	Days Inland	Days Coastal	Days Inland	Days Coastal	Days Inland	Days Coastal
Jan	5	26	3	29	8	23
Feb	0	28	4	24	0	28
March	2	29	7	24	2	29
April	0	30	13	17	0	30
May	2	29	26	5	0	31
June	14	16	26	5	12	18
July	18	13	24	7	17	14
Aug	17	15	17	15	17	14
Sep	20	10	19	11	17	13
Oct	12	19	14	17	8	24
Nov	5	25	13	17	7	23
Dec	1	30	8	23	10	21

[1]Hanson and Emmons report sightings in inland waters. For purposes of this consultation analysis, and because the population is highly visible when in inland waters, NMFS assumes that when not sighted in inland waters the whales are in their coastal range.

Late summer and early fall movements of Southern Residents in the Georgia Basin are consistent, with strong site fidelity shown to the region as a whole and high occurrence in the San Juan Island area (Hanson and Emmons 2010; Hauser et al. 2007). There is inter-annual variability in arrival time and days present in inland waters from spring through fall, with late arrivals and fewer days present during spring in recent years potentially related to weak returns of spring and early summer Chinook to the Fraser River (Hanson and Emmons 2010). Similarly, recent high occurrence in late summer may relate to greater than average Chinook returns to South Thompson tributary of the Fraser River (Hanson and Emmons 2010). During fall and early winter, Southern Resident pods, and J pod in particular, expand their routine movements into Puget Sound, likely to take advantage of chum and Chinook salmon runs (Hanson et al. 2010a; Osborne 1999). During late fall, winter, and early spring, the ranges and movements of the Southern Residents are less known. Sightings through the Strait of Juan de Fuca in late fall suggest that activity shifts to the outer coasts of Vancouver Island and Washington (Krahn et al. 2002).

The Southern Resident killer whales were formerly thought to range southward along the coast to about Grays Harbor (Bigg et al. 1990) or the mouth of the Columbia River (Ford et al. 2000). However, recent sightings of members of K and L pods in Oregon and California have considerably extended the southern limit of their known range (NMFS 2008a). There have been verified visual sightings or strandings of J, K or L pods along the outer coast from 1975 to present with most made from January through April (summarized in NMFS 2008a, and NWFSC unpubl. data). A subset of these sightings include 16 records off Vancouver Island and the Queen Charlottes, 12 off Washington, 4 off Oregon, and 12 off central California. Most records have occurred since 1996, but this may be because of increased viewing effort along the coast for this time of year.

Sightings in Monterey Bay, California coincided with occurrence of salmon, with feeding witnessed in 2000 (Black et al. 2001). Southern Residents were also sighted in Monterey Bay during 2008, when salmon runs from California were expected to be near record lows (PFMC 2010). L pod was also seen feeding on unidentified salmon off Westport, Washington, in March 2004 during the spring Chinook run in the Columbia River (M. B. Hanson, personal observation, as cited in Krahn et al. 2004). In March, 2005 L pod was sighted working a circuit across the Columbia River plume from the North Jetty across to the South Jetty during the spring Chinook run in the Columbia River (Zamon et al. 2007). Also in March of 2006, K and L pods were encountered off the Columbia River (Hanson et al. 2008). L pod was again seen feeding off Westport, Washington in March 2009, and genetic analysis of prey remains collected from two predation events identified one fish as spring Chinook and the other as a summer/fall Chinook from Columbia River stocks (Hanson et al 2010b).

The Northwest Fisheries Science Center (NWFSC) also deploys and collects data from remote autonomous acoustic recorders in coastal waters of Washington State, and in 2009 alone documented 52 Southern Resident killer whale detections from this acoustic system (Emmons et al. 2009). The Department of Fisheries and Oceans (DFO), Canada also maintains acoustic recorders in British Columbia. When the NWFSC and DFO analyze these data, more information will be available about the seasonal distribution, movements and habitat use of Southern Residents, specifically in coastal waters off Washington and British Columbia.

Limiting Factors and Threats. Several factors identified in the final recovery plan for Southern Resident killer whales may be limiting recovery. These are quantity and quality of prey, exposure to toxic chemicals that accumulate in top predators, disturbance from sound and vessels. Oil spills are also a risk factor. It is likely that multiple threats are acting in concert to impact the whales. Although it is not clear which threat or threats are most significant to the survival and recovery of Southern Residents, all of the threats identified are potential limiting factors in their population dynamics (NMFS 2008a). Here we focus on toxic chemicals and prey because these are affected by the proposed action. The discussion in the Environmental Baseline and Cumulative Effects sections contain a thorough evaluation of all threats in the action area.

Toxic Chemicals. Contaminants enter fresh and marine waters and sediments from numerous sources such as atmospheric transport and deposition, ocean current transport, and terrestrial runoff (Iwata et al. 1993; Grant and Ross 2002; Hartwell 2004), but are typically concentrated near populated areas of high human activity and industrialization. Oceans act as a repository for domestic and industrial wastes and significant contaminant concentrations have been measured in the sediment, water and biota. Persistent pollutants can biomagnify or accumulate up the food chain in such a degree where levels in upper trophic-level species can have significantly higher concentrations than that found in the water column or in lower trophic-level species. Southern Resident killer whales are exposed to relatively high levels of persistent pollutants because they are long-lived, upper trophic-level predators that are in close proximity to industrial and agricultural areas. Consequentially, Southern Residents are a highly contaminated whale population.

Persistent pollutants can be highly lipophilic (i.e., fat soluble) and are primarily stored in the fatty tissues in marine mammals (O'Shea 1999; Reijnders and Aguilar 2002). Therefore, when killer whales consume contaminated prey they store the contaminants primarily in their blubber. Persistent pollutants can resist metabolic degradation and can remain stored in the blubber of an individual for extended periods of time. When prey is scarce and when other stressors reduce foraging efficiency (e.g., as possible from vessel disturbance, disease, etc.), killer whales metabolize their blubber lipid stores and the contaminants can become mobilized to other organs and or they can remain in the blubber and become more concentrated (Krahn et al. 2002). Nursing females can also transmit large quantities of contaminants to their offspring, particularly during lactation. The mobilized contaminants can reduce the whales' resistance to disease, can affect reproduction, disrupt the endocrine system, disrupt enzyme function and vitamin A physiology, induce developmental neurotoxicity, and cause skeletal deformities (see NMFS 2008a for a review).

There are several contaminants of concern that have been highlighted in the Southern Resident killer whale Recovery Plan (Table 3). Some of these contaminants do not need to be in high concentration in a species to be toxic and have long been recognized as problematic for the Southern Resident killer whales. The organochlorines (e.g. PCBs and DDTs) are thought to pose the greatest risk to killer whales (Ross et al. 2000; Center for Biological Diversity 2001; Krahn et al 2002). PCBs were designed for chemical stability and were historically used in paints and sealants, industrial lubricants and coolants, and flame-retardants. DDTs were primarily used to control insects in commercial and agricultural areas, forests, homes, and gardens. PCBs and DDTs were banned in the 1970s and 1980s due to their toxicity in humans and wildlife. Although levels of PCBs and DDTs have dramatically decreased in environmental samples since the mid 1970s (Mearns et al. 1988; Lieberg-Clark et al. 1995; Calambokidis et al. 2001; Rigét et al. 2010), these compounds continue to be measured in marine biota around the world, including killer whales and their prey.

The majority of Southern Residents have high levels of PCBs (Ross et al. 2000; Krahn et al. 2007, 2009) that exceed a health-effects threshold (17,000 ng/g lipid) derived by Kannan et al. (2000) and Ross et al. (1996) for PCBs in marine mammal blubber. The PCB health-effects threshold is associated with reduced immune function and reproductive failure in harbor seals (Reijnders 1986; de Swart et al. 1994; Ross et al. 1996; Kannan et al. 2000). Hickie et al. (2007) projected that it will take at least 50 years for the Southern Residents to drop below the threshold. Moreover, juvenile Southern Resident killer whales have blubber concentrations that are currently 2 to 3.6 times higher than the established health-effects threshold (Krahn et al. 2009). Similarly, Southern Residents also have high levels of measured DDTs in their blubber (Krahn et al. 2007, 2009).

Recent studies suggest that certain pharmaceuticals and personal care products (PPCPs) may also accumulate in killer whales. Synthetic musks and antibacterial chemicals (e.g. Triclosan) have been detected in dolphins and porpoises in coastal waters off Japan and the southeastern United States and in harbor seals off the California Coast (Fair et al. 2009; Kannan et al. 2005; Nakata 2005; Nakata et al. 2007). A wider range of PPCPs, including anti-depressants, cholesterol lowering drugs, antihistamines, and drugs affecting blood pressure and cholesterol levels have been detected in tissues of fish from urban areas and sites near wastewater treatment plants (Brooks et al. 2005; Ramirez et al 2009), suggesting possible contamination of prey. As yet we have no data on concentrations of PPCPs in either killer whales or their prey species, but they could be a concern because of their widespread occurrence, potential for biomagnification, and biological activity.

Recent decades have brought rising concern over a list of the so-called “emerging” contaminants and other pollutants, such as the PBDEs. PBDEs have been used as additive flame-retardants in many products including electronics, textiles and plastics. Additive flame-retardants can readily dissociate from the products they are added to and discharge into the environment. Due to the increase in fire regulations in many countries, the use of PBDEs has increased in the last few decades. PBDEs have been identified as a growing concern and have a ubiquitous distribution with increasing levels found in various matrices including surface water, sewage sludge, sediment, air, and biota (Hale et al. 2003; Hites 2004). PBDEs are structurally comparable to PCBs and share some similar toxicological properties (Hooper and McDonald 2000). Manufacturers of two commercial forms of PBDEs, penta-BDE and octa-BDE, agreed to voluntarily stop producing these two forms by the end of 2004. In January 2006, the Washington State Department of Ecology (DOE) and the Washington State Department of Health (DOH) issued a Final PBDE Chemical Action Plan (WDOE and WDOH 2006) that recommended the Legislature prohibit the three main types of PBDEs used in consumer products (e.g., penta-, octa- and deca-BDEs). The penta and octa forms are currently being phased out in Washington State following a bill (ESHB1024) that was passed in 2007. This bill banned the use of the penta and octa forms by 2008, banned the use of the deca form in mattresses by 2008, and banned the use of the deca form in televisions, computers, and furniture by 2011.

Table 3. Persistent pollutants that may pose a risk to resident killer whales. (*Source: updated from NMFS 2008a*).[1]

Pollutant	Use/Source	Persistent	Bio-accumulate	Risk
DDT (Dichlorodi-phenyl trichloroethane	pesticide used in some countries, banned in North America, persists in terrestrial runoff 30 years post ban, enters atmosphere from areas where still in use	yes	yes	Reproductive impairment, immunosuppression, adrenal and thyroid effects
PCBs Polychlorinated Biphenyls	electrical transformer and capacitor fluid, limited use in North America but enters environment from runoff, spills and incineration	yes	yes	reproductive impairment, skeletal abnormalities, immunotoxicity, carcinogenic, and endocrine disruption
Dioxins and Furans	by-product of chlorine bleaching, wood product processing and incomplete combustion. Mills less of a source now. Current sources include burning of salt-laden wood, municipal incinerators, and residential wood and wood waste combustion, in runoff from sewage sludge, wood treatment	yes	yes	thymus and liver damage, birth defects, reproductive impairment, endocrine disruption, immunotoxicity and cancer
PAHs Persistent Polycyclic aromatic hydrocarbons	by-product of fuel combustion, aluminum smelting, wood treatment, oil spills, metallurgical and coking plants, pulp and paper mills	yes	no	Carcinogenic and cardiac dysfunction, developmental neurotoxicity, reproductive dysfunction, immunotoxicity, and eggshell thinning
flame retardants, esp. PBBs and PBDEs Polybrominated diphenyl ethers	flame retardants; in electrical components and backings of televisions and computers, in textiles and vehicle seats, ubiquitous in environment. 2/3 product PBDEs banned in Europe. Same two products withdrawn from North American marketplace in 2005, but one (deca) product still used globally.	yes	yes	endocrine disruption, impairs liver and thyroid
PFOs Perfluro-octane sulfonate	stain, water and oil repellent (included in Scotchgard until recently), fire fighting foam, fire retardants, insecticides and refrigerants, ubiquitous in environment	yes	yes but in blood, liver, kidney and muscle	promotes tumor growth
TBT, DBT Tributyltin Dibutyltin	antifoulant pesticide used on vessels	yes	yes	unknown but recently associated with hearing loss
PCPs (Polychlorinated paraffins)	flame retardants, plasticizers, paints, sealants and additives in lubricating oils	yes	yes	endocrine disruption
PCNs Polychlorinated napthalenes	ship insulation, electrical wires and capacitors, engine oil additive, municipal waste incineration and chlor-alkali plants, contaminant in PCBs	yes	yes	endocrine disruption
APEs Alkyl-phenol ethoxylates	detergents, shampoos, paints, pesticides, plastics, pulp and paper mills, textile industry found in sewage effluent and sediments	moderate	moderate	endocrine disruption
PCTs Polychlorinated terphenyls	fire retardants, plasticizers, lubricants, inks and sealants, enters environment in runoff	yes	yes	endocrine disruption and reproductive impairment

[1]Grant and Ross 2002, Lindstrom et al. 1999, Hooper and MacDonald 2000, Kannan et al. 2001,

Hall et al. 2003; Legler and Brower 2003, Van de Vijver et al. 2003, Rayne et al. 2004, Song et al. 2005, Darnerud 2008, Legler 2008, Fernie et al. 2009, Kodavanti et al. 2010.

Although specific regional data is limited for PBDE levels, the environmental levels of a few PBDE congeners appear to have surpassed PCBs in some areas in North America (Hale et al. 2003; Ross et al. 2009). Recent studies have documented relatively high concentrations of PBDEs in Southern Resident killer whales (Krahn et al. 2007, 2009; Mongillo 2009). Although PBDE levels in the whales are lower than PCBs or DDTs (Krahn et al. 2007, 2009) concern is growing because PBDE exposure and accumulation will likely continue in the future increasing the risk to the health of the killer whales. Several other marine species have recently experienced an almost exponential increase in PBDE concentrations (e.g., Ikonomou et al. 2002; Lebeuf et al. 2004).

Contaminant-Induced Health Effects from Exposure to PBDEs. There is currently no PBDE health-effects threshold identified for killer whales. Effects due to PBDE exposure may potentially be species-specific, dose-dependent, and congener-specific. For example, the heavier weighted congeners appear to be less potent than the lighter weighted congeners (see Darnerud et al. 2001). However, Southern Resident killer whales have higher PBDE concentrations (Krahn et al. 2007, 2009) than those associated with altered thyroid hormone levels in post-weaned and juvenile grey seals (Hall et al. 2003). In fact, one juvenile killer whale had a PBDE blubber concentration (Krahn et al. 2007) that was 10 times higher than those associated with the endocrine disruption in grey seals (Hall et al. 2003;).

The PBDEs are potential endocrine disruptors that can affect thyroid hormone levels, and can cause subtle neurobehavioral effects and reproductive effects in numerous species both *in vivo* and *in vitro* (Legler and Brouwer 2003; Darnerud 2008; Legler 2008; Kodavanti et al. 2010). For example, some PBDE metabolites are structurally similar to thyroid hormones and these metabolites can disrupt the thyroid hormone homeostasis in mice and rats (Zhou et al. 2001; Zhou et al. 2002; Richardson et al. 2008). Disruption of hormone homeostasis can cause developmental neurotoxicity. The metabolites can compete or displace thyroid hormones and bind to thyroid hormone receptors, alter gene expression, reduce the transfer of retinol and T_4 (a thyroid hormone) to target organs, and decrease the availability of progesterone (Meerts et al. 2000; Houde et al. 2005; Boas et al. 2006). The potency of PBDEs to compete with T_4 is congener-specific and metabolic enzyme-specific (Meerts et al. 2000). For example, parent PBDEs were not shown to bind to human transthyretin *in vitro*, however the hydroxylated PBDEs were capable of binding to the receptors (Meerts et al. 2000).

Binding to thyroid receptors not only disrupts the transport of the hormones essential for brain development, but also transports PBDEs across the blood-brain and placental barriers (de Boer et al. 2000). PBDE exposure can cause continuing behavioral alterations, and reduced learning and memory (Costa and Giordano 2007). Furthermore, young individuals may not be able to excrete PBDEs as efficiently as older individuals and may accumulate higher concentrations in the brain (Costa and Giordano 2007). The capacity for metabolic breakdown of PBDEs may also increase with age or with increasing concentrations (Weijs et al. 2009). It may also be possible that a contaminant-induced reduction of thyroid hormone levels has the potential to alter hearing and communication in mammals. For example, a 50-60 percent reduction of the thyroid hormone T_4 in rats during the postnatal period correlated with hearing loss in adults (Crofton 2004).

Endocrine disruptors can mimic or offset reproductive processes. Consequently, adverse reproductive effects have been associated with PBDE exposure. For example, sweet preference is a sexually dimorphic behavior in rats. Typically adult males consume less sweets (or sweetened solutions) than females. Exposure to the congener BDE-99 demonstrated an increase in sweet preference, indicating behavioral feminization (Hany et al. 1999, Lilienthal et al. 2006). Reproductive toxicity has also been reported in male rats exposed to low doses of BDE-99 by permanently impairing spermatogenesis in adult rats including reductions in sperm and spermatid counts as well as smaller testes (Kuriyama et al. 2005). In female offspring, a delay in the onset of puberty was observed, and a reduction in the number of ovarian follicle numbers (Lilienthal et al. 2006).

The timing of PBDE exposure can affect the degree of toxicity. The maturation and development of the central nervous system has two main stages. The first stage consists of early embryonic brain development, whereas the second stage is referred to as the brain growth spurt. The most critical or sensitive period for developmental neurotoxicity appears to occur during the height of the brain growth spurt. For example, neonatal mice exposed to BDE-99 during a critical period of brain development experienced impaired spontaneous behavior (i.e., behavior important for survival such as hunting and predator avoidance), however, mice exposed after the growth spurt did not experience the neurotoxic effects (Eriksson et al. 2002). This study indicates that adverse health effects are not only dose-dependent and species-specific, but the timing of exposure is a significant factor. Other studies where animals are exposed to PBDEs during the defined critical period have shown to cause reductions in sperm and spermatid counts in adult rats and increase hyperactivity in their offspring, cause morphological effects in the thyroid, liver, and kidneys, increase circulating thyroid hormones, and alter spontaneous behavior (Viberg et al 2003, 2007; Kuriyama et al. 2005). Additionally, neonatal exposure may produce long-term modifications to the cholinergic or neurotransmitter system (Talsness 2008). Therefore, killer whale calves are likely more susceptible to adverse health effects than killer whales only exposed as adults because they are exposed to contaminants during the critical period of development. The influx of toxins in killer whale calves is a cause for concern because the growth and development of an individual is highly dependent on normal levels of thyroid hormones (Boas et al. 2006). PBDEs may disrupt normal hormone function by altering the concentrations of circulating thyroid hormone (e.g. Hall et al. 2003) as well as interfere with developmental processes (Eriksson et al. 2002, 2006).

While PBDEs can present direct health threats to hormonal regulation, neural development and function, and reproduction as discussed above, they can also alter susceptibilities to infectious diseases. One mechanism of action of inducing contaminant effects is through interactions with the aryl hydrocarbon receptor (AhR), generally described as “dioxin-like” effects. “Dioxin-like” contaminants are particularly effective at immunotoxicity across a range of species. PCBs, PBDEs, and DDTs have well documented effects on the immune system in a wide range of experimental animals (e.g., Thomas and Hinsdill 1978; Thomas and Hinsdill 1980; Safe et al 1989; Dahlman et al. 1994). Killer whales are exposed to a large array of pathogens in the aquatic environment. In the absence of a robust immune system, the individual whale’s health, or its ability to endure and thrive, can become compromised. The immune system is important in patrolling and eliminating cells that undergo malignant transformation. If this immune surveillance is compromised the potential exists for tumors to develop. For example, St.

Lawrence belugas had a high occurrence of tumors and lesions, and some evidence of immunosuppression, and have high PCB concentrations (Béland et al. 1993, Martineau et al. 1994). California sea lions that died of carcinoma had higher PCB concentrations compared to California sea lions that died without carcinoma (Ylitalo et al. 2005). Contaminants may play a role in the development of disease by suppressing the immune system or through genotoxic mutation and tumor promotion (Ylitalo et al. 2005).

Mixture Effects and Non-Linear Dose-Response Curves. Southern Resident killer whales are exposed to a number of toxic chemicals in the Puget Sound and the interactions of these chemicals have the potential to be additive (when the effects from two or more chemicals equal the sum of the effects of the isolated chemicals), synergistic (where the effects from the interaction is greater than the sum of the effects of the isolated chemicals), or antagonistic (where the effects from the interaction is less than the sum of the effects from the isolated chemicals). Recent evidence suggests that PBDEs interact with PCBs synergistically in laboratory species and enhance toxicity (e.g. Eriksson et al. 2006; He et al. 2009a; He et al. 2010b; He et al. 2010). Disregarding synergistic interactions between persistent pollutants that are currently found in high concentrations in the Southern Resident killer whales may underestimate risk to an individual or to the entire DPS. Because killer whale calves and adult males have relatively high contaminant concentration levels, and pregnant or lactating female killer whales mobilize their lipids (and lipophilic contaminants) into circulation from transplacental transfer and lactation, they are more susceptible to enhanced detrimental biological health effects resulting from the additive and synergistic interactions of multiple contaminants.

Although health risks are probably elevated as a result of interactions between toxic chemicals, and wildlife is rarely exposed to single compounds, the majority of studies have examined the effects of isolated chemicals. It has only been in more recent years that studies have examined health effects from exposure to mixtures of chemicals. For example, a few recent studies have highlighted the importance of evaluating mixture effects (Hallgren and Darnerud 2002; Crofton et al. 2005; Eriksson et al. 2006; Fischer 2008; He et al. 2009a, b, 2010). Mixture effects case studies that have examined effects from the interaction of PBDEs and PCBs (e.g. Eriksson et al. 2006; He et al. 2009 a,b; He et al. 2010) demonstrate that exposure to the chemical mixture at a critical developmental growth period result in enhanced toxicity and that the defects worsen with age.

The practice of examining only high doses of contaminants, especially endocrine disruptors such as PBDEs, may underestimate risk (for a review, see Welshons et al. 2003) because some contaminants can interact at doses below the no observed effect concentrations (NOECs) and produce significant effects (Silva et al. 2002). For example, Crofton et al. (2005) tested the hypothesis that a mixture of thyroid hormone-disrupting chemicals has additive dose-response effects. They demonstrated that the effects from a mixture consisting of thyroid hormone disrupters can be additive at low doses and synergistic at high doses and more importantly, the highest mixture dose levels were at or below the NOECs of the chemicals. Endocrine disruptors, when isolated, have shown to produce nonlinear (e.g., U-shaped or J shaped) dose-response curves. For example, PBDE concentrations in the blubber of grey seals significantly contributed to circulating thyroid hormone concentrations (Hall et al. 2003). They found a positive association between PBDEs and circulating thyroid hormones, in contrast to several laboratory studies that have reported a negative correlation. Furthermore, the PBDE concentrations in the grey seals were at much lower doses than were used in laboratory studies, suggesting a hormetic dose-response (or an enhancement of the response at low doses and an inhibition at high doses). A nonmonotonic dose-response relationship is not uncommon in the literature. In a separate study, cell proliferation was observed when cells were treated with 17β-Estradiol and low concentrations of PCB congeners 138, 153, and 180, whereas inhibited cell growth was observed at high concentrations of these PCB congeners (Bonefeld-Jørgensen et al. 2001).

Additive or synergistic mixture effects can occur from a wide range of doses; therefore, even low concentrations of persistent pollutants when combined together have the potential to cause adverse health effects in the Southern Resident killer whales. Although it is not clear if contaminant levels in the Southern Residents are at or near a health-effects threshold, it is reasonable to assume that a combination of their current high PCB concentrations and their increasing PBDE concentrations has a potential to disrupt the reproductive system, the endocrine system, and the immune system within a whales' lifetime.

Prey. Healthy killer whale populations depend on adequate prey levels. First, we discuss the prey requirements of Southern Residents followed by an assessment of threats to the quantity and quality of their prey.

Prey Requirements. Southern Resident killer whales consume a variety of fish species (22 species) and one species of squid (Scheffer and Slipp 1948; Ford et al. 1998, 2000; Ford and Ellis 2006; Saulitis et al. 2000; Hanson et al. 2010c), but salmon are identified as their primary prey (i.e., a high percent of prey consumed during spring, summer, and fall, from long-term studies of resident killer whale diet; Ford and Ellis 2006; Hanson et al. 2010c). Feeding records for Southern and Northern Residents show a predominant consumption of Chinook salmon during late spring to fall (Ford and Ellis 2006). Chum salmon are also taken in significant amounts, especially in fall. Other salmon eaten include coho, pink, steelhead (*O. mykiss*), and sockeye (*O. nerka*). The non salmonids include Pacific herring, sablefish, Pacific halibut, quillback and yelloweye rockfish (*Sebastes maliger*), lingcod (*Ophiodon elongates*), and Dover sole (*Microstomus pacificus*) (Ford et al. 1998; Hanson et al. 2010c). Chinook were the primary prey despite the much lower abundance of Chinook in the study area in comparison to other salmonids (primarily sockeye), for mechanisms that remain unknown but factors of potential importance include the species' large size, high fat and energy content, and year-round occurrence in the area. Killer whales also captured older (i.e., larger) than average Chinook (Ford and Ellis 2006). Recent research suggests that killer whales are capable of detecting, localizing, and recognizing Chinook salmon through their ability to distinguish Chinook echo structure as different from other salmon (Au et al. 2010).

Southern Residents are the subject of ongoing research, including direct observation, scale and tissue sampling of prey remains, and fecal sampling. A recent publication by Hanson et al. (2010c) provides the best available scientific information on diet composition of Southern Residents in inland waters during summer months. The results provide information on (1) the percentage of Chinook in the whales' diet, and (2) the predominant river of origin of those Chinook. Other research and analysis provides additional information on the age of prey consumed (Hanson, unpubl. data, as summarized in Ward et al. 2010), indicating that the whales are consuming mostly larger (i.e., older) Chinook.

Scale and tissue sampling in inland waters from May to September indicate that the Southern Residents' diet consists of a high percentage of Chinook, with an overall average of 88 percent Chinook across the timeframe and monthly proportions as high as >90% Chinook (i.e., July: 98 percent and August: 92 percent, see S/T sample type in Table 2 Hanson et al. 2010c). Fecal samples are also considered in Hanson et al. (2010c) but are not used to estimate proportion of the Southern Residents' diet, because the data from these samples represents presence or absence of prey species, but not proportion of diet. DNA quantification methods can be used to estimate the proportion of diet from fecal samples (i.e., Deagle et al. 2005). This technique is still in the developmental stages. However, preliminary DNA quantification results from Hanson et al. (2010c) samples indicate that Chinook make up the bulk of the prey DNA in the fecal samples (Ford et al. 2011b).

Genetic analysis of the Hanson et al. (2010c) samples indicate that when Southern Resident killer whales are in inland waters from May to September, they consume Chinook stocks that originate from regions including the Fraser River (including Upper Fraser, Mid Fraser, Lower Fraser, N. Thompson, S. Thompson and Lower Thompson), Puget Sound (N. and S. Puget Sound), the Central British Columbia Coast and West and East Vancouver Island. Hanson et al. (2010c) find that the whales are likely consuming Chinook salmon stocks at least roughly proportional to their local abundance, as inferred by Chinook run-timing pattern and the stocks represented in killer whale prey for a specific area of inland waters, the San Juan Islands. Ongoing studies also confirm a shift to chum salmon in fall (Ford et al. 2010a; Hanson et al. 2010a).

Although less is known about the diet of Southern Residents off the Pacific coast, the available information indicates that salmon, and Chinook in particular, are also important when the whales occur in coastal waters. To date, there are direct observations of two different predation events (where the prey was identified to species and stock from genetic analysis of prey remains) when the whales were in coastal waters. Both were identified as Columbia River Chinook stocks (Hanson et al. 2010b). Chemical analyses also support the importance of salmon in the year round diet of Southern Resident killer whales (Krahn et al. 2002, 2007a, 2009). Krahn et al. (2002), examined the ratios of DDT (and its metabolites) to various PCB compounds in the whales, and concluded that the whales feed primarily on salmon throughout the year rather than other fish species. Krahn et al. (2007a) analyzed stable isotopes from tissue samples collected in 1996 and 2004/2006. Carbon and nitrogen stable isotopes indicated that J and L pods consumed prey from similar trophic levels in 2004/2006 and showed no evidence of a large shift in the trophic level of prey consumed by L pod between 1996 and 2004/2006. The predominance of Chinook in their diet in inland waters, even when other species are more abundant, combined with information to date about prey in coastal waters (above), makes it reasonable to expect that Chinook salmon is equally predominant in the whales' diet when available in coastal waters. It is also reasonable to expect that the diet of Southern Residents is predominantly larger Chinook when available in coastal waters. The diet of Southern Residents in coastal waters is a subject of ongoing research.

Quantity of Prey. Human influences have had profound impacts on the abundance of many prey species in the northeastern Pacific during the past 150 years, including salmon. The health and abundance of wild salmon stocks have been negatively affected by altered or degraded freshwater and estuarine habitat, including numerous land use activities, from hydropower systems to urbanization, forestry, agriculture and development. Harmful artificial propagation practices and overfishing have also negatively affected wild salmon stocks. Sections 2.2.1 and 2.3.1 provide a comprehensive overview of limiting factors for Puget Sound Chinook, as does the Puget Sound Salmon Recovery Plan (Shared Strategy 2007 and NMFS 2007). Predation also contributes to natural mortality of salmon. Salmonids are prey for pelagic fish, birds, and marine mammals including killer whales.

While wild salmon stocks have declined in many areas, hatchery production has supplemented additional prey. Currently, hatchery production contributes a significant component of the salmon prey base returning to watersheds within the range of Southern Resident killer whales (i.e., review PFMC 2011 for Puget Sound, Barnett-Johnson et al. 2007 for Central Valley California, and NMFS 2008b for Columbia River Basin). Although hatchery production has contributed some offset of the historical declines in the abundance of wild salmon within the range of Southern Residents, hatcheries also pose risks to wild salmon populations (i.e., Ford 2002; Nickelson et al. 1986; Levin and Williams 2002; Naish et al. 2008). In recent decades, managers have been moving toward hatchery reform, and are in the process of reducing risks identified in hatchery programs, through region-wide recovery planning efforts and hatchery program reviews. Healthy wild salmon populations are important to the long-term maintenance of prey populations available to Southern Resident killer whales, because it is uncertain whether a hatchery dominated mix of stocks is sustainable indefinitely.

Salmon abundance is also substantially affected by climate variability in freshwater and marine environments, particularly by conditions during early life-history stages of salmon (NMFS 2008b). Sources of variability include inter-annual climatic variations (e.g., El Niño and LaNiña), longer term cycles in ocean conditions (e.g., Pacific Decadal Oscillation, Mantua et al. 1997), and ongoing global climate change. For example, climate variability can affect ocean productivity in the marine environment and water storage (e.g. snow pack) and in-stream flow in the freshwater environment. Early life-stage growth and survival of salmon can be negatively affected when climate variability results in conditions that hinder ocean productivity (e.g., Scheurell and Williams 2005) and/or water storage (e.g., ISAB 2007) in marine and freshwater systems, respectively. Severe flooding in freshwater systems can also constrain salmon populations (NMFS 2008c). The availability of adult salmon may be reduced in years following unfavorable conditions to the early life-stage growth and survival of salmon.

When prey is scarce, whales likely spend more time foraging than when it is plentiful. Increased energy expenditure and prey limitation can cause nutritional stress. Nutritional stress is the condition of being unable to acquire adequate energy and nutrients from prey resources and as a chronic condition can lead to reduced body size and condition of individuals and lower reproductive and survival rates of a population (e.g., Trites and Donnelly 2003). The Center for Whale Research has observed the very poor body condition in 13 members of the Southern Resident population, and all but two of those whales subsequently died (Durban et al. 2009). Both females and males across a range of ages were found in poor body condition (Durban et al. 2009). Food scarcity could also cause whales to draw on fat stores, mobilizing contaminants stored in their fat that are at relatively high levels (Krahn et al. 2007, 2009; Mongillo 2009) and affecting reproduction and immune function (as discussed above).

Quality of Prey. The quality of Chinook salmon, Southern Resident killer whales' primary prey, is likely influenced by a variety of factors, including size of the fish, their fat content, contaminant load, and origin (natural vs. hatchery). Overall, Chinook have the highest lipid content (Stanby 1976; Winship and Trites 2003), largest size, and highest caloric value per kg of any salmonid species (Ford and Ellis 2006; Osborne 1999). Details about contaminant load, size, and origin are provided below.

Contaminant Load. Various studies have documented a range of concentrations of persistent organic pollutants (POPs) in many populations of adult Pacific salmon (refer to Table 4). Based on the data summarized in Table 4, POP accumulation in Pacific salmon is primarily determined by geographic proximity to contaminated environments (Mongillo et al. in prep). Because Chinook salmon are distributed in more coastal waters, they are more readily exposed to contaminants that are present in coastal waters than other species. In contrast, sockeye, pink, and chum salmon have lower POP concentrations because by the end of their first year, they have migrated through the coastal waters and are found in the open waters of the North Pacific, Gulf of Alaska, and Bering Sea (Quinn 2005). Measured average concentrations of PCBs and PBDEs were highest for Chinook (29 ng/g and 6.22 ng/g, respectively) intermediate for coho (14 ng/g and 0.20 ng/g, respectively), less for sockeye (7.6 ng/g and 0.15 ng/g), and lowest for pink (2.4 ng/g and 0.18 ng/g, respectively) and chum salmon (2.6 ng/g and 0.14 ng/g, respectively [see Table 4]). Similarly, average DDT values were higher in Chinook and coho salmon compared to sockeye and lowest for pink and chum salmon (see Table 4). Highest levels of PCBs and PBDEs were measured in Puget Sound populations, intermediate levels were measured in California and Oregon populations, and the lowest average levels were measured in populations off Alaska (Mongillo et al. in prep). The biological traits in Pacific salmon (e.g. trophic status, lipid content, age, exposure duration, metabolism, and detoxification) may also affect the degree to which POPs accumulate (Mongillo et al. in prep).

Size. Size of individual salmon is an aspect of prey quality that could affect the foraging efficiency of Southern Resident killer whales. As discussed above, available data suggests that Southern Residents consume larger prey. The degree to which this is a function of the availability of all sizes of fish in the summer range of the whales, their ability to detect all sizes or a true preference for only large fish is unknown. It is possible although not conclusive that there has been a historical decrease in salmon age, size, or size at a given age (i.e., Bigler et al. 1996, but also see PFMC data (PFMC 2011). Fish size is influenced by factors such as environmental conditions, selectivity in fishing effort through gear type, fishing season or regulations, and hatchery practices. The available information on size is also confounded by factors including inter-population difference, when the size was recorded, and differing data sources and sampling methods (review in Quinn 2005).

Origin. Southern Resident killer whales likely consume both natural and hatchery salmon (Hanson et al. 2010c). The best available information does not indicate that natural and hatchery salmon generally differ in size, run-timing, or ocean distribution (e.g., Nickum et al. 2004; NMFS 2008b; Weitkamp and Neely 2002), which are differences that could affect Southern Residents. However, there is evidence of size and run-timing differences between hatchery and natural salmon from specific river systems or runs (i.e., size and run timing differences as described for Willamette River Chinook in NMFS 2008d). Potential run-specific differences in the quality of natural and hatchery salmon are evaluated where data are available.

Table 4. Lipid and persistent organic pollutant concentrations (ng/g wet weight) of adult and subadult Pacific salmon sampled in terminal areas. Terminal areas include coastal marine water and river mouths through which fish migrate *en route* to their natal stream. From Mongillo et al. (in prep).

Species	Region	sub-region	Population	n	Tissue Analyzed	Lipid (%)	PCBs	DDTs	PBDEs	Citation
Chinook salmon	Alaska	Unknown	unknown	2	muscle w/o skin	NR	5.6	NR	0.95	4
	Alaska	Aleutian Islands	unknown	3	muscle w/skin	7.6	5.0	22	0.71	13, 14*
	Alaska	SE Alaska/ Gulf of Alaska/ Bering Sea	unknown	35	muscle w/o skin	9.7	11	7.1	0.53	20
	Alaska	SE Alaska	unknown	3	muscle w/skin	NR	8.0	NR	0.50	5*, 6*
	Alaska	South Central	River	10	muscle w/o skin	NR	9.1	9.8	NR	12
		Alaskan Chinook salmon Average				8.7	7.7	13.0	0.67	
	British Columbia	BC North Coast	Skeena	30	whole body	NR	7.3	7.3	0.08	10
	British Columbia	Fraser River	Thompson	6	muscle w/o skin	10	9.1	1.5	NR	1
	British Columbia	Fraser River		13	whole body	NR	9.4	6.6	0.80	10
	British Columbia	Fraser River	Thompson	7	muscle w/o skin	12	8.6	7.7	1.54	16**
	British Columbia	Fraser River	Shuswap	2	muscle w/o skin	3.0	9.8	5.5	NR	16**
	British Columbia	Fraser River	Harrison	6	muscle w/o skin	5.4	47	4.3	17.7	1
		Fraser River Chinook salmon Average (excluding Harrison)				8.3	10	5.7	1.67	
		British Columbia Chinook salmon Average				7.6	15	5.5	4.87	
	Washington	Puget Sound	Nooksack River	28	muscle w/o skin	3.5	37	NR	NR	11

	Washington	Puget Sound	Skagit River	29	muscle w/o skin	4.8	40	NR	NR	11
	Washington	Puget Sound	Duwamish River	65	muscle w/o skin	7.3	56	NR	NR	11
	Washington	Puget Sound	Nisqually River	20	muscle w/o skin	3.8	41	NR	NR	11
	Washington	Puget Sound	Deschutes River	34	muscle w/o skin	1.7	59	NR	NR	11
	Washington	Puget Sound	PS mixed	28	muscle w/o skin	4.8	76	NR	NR	11
	Washington	Puget Sound	Duwamish River	3	whole body	6.4	35	18.3	6.43	1
	Washington	Puget Sound	Deschutes River	4	whole body	4.3	56	NR	NR	1
	Washington	Puget Sound	Deschutes River	10	muscle w/o skin	1.0	49	NR	NR	8
	Washington	Puget Sound	Issaquah Creek	10	muscle w/o skin	0.6	49	NR	NR	8
	Washington	Puget Sound	PS mixed	36	whole body	NR	43	29.1	18.9	10
	Washington	Puget Sound	PS mixed	34	whole body	NR	91	16.4	42.2	10
	Washington	WA Coast	Makah	10	muscle w/o skin	1.5	19	NR	NR	8
	Washington	WA Coast	Quinault	10	muscle w/o skin	1.8	16	NR	NR	8
		Puget Sound Chinook salmon Average				3.8	53	21.3	22.5	
		Washington Coast Chinook salmon Average				1.7	17	NR	NR	
		Washington Chinook salmon Average				3.5	48	21.3	22.5	
	Oregon	Unknown	unknown	3	muscle w/skin	NR	10	NR	2.10	5*, 6*
	Oregon	Columbia River	unknown Fall	17	whole body	NR	18	19.9	3.69	10
	Oregon	Columbia River	unknown Spring	20	whole body	NR	33	34.8	9.77	10
	Oregon	Columbia River	mixed fall Chinook	15	muscle w/skin	7.0	37	21.0	NR	17
	Oregon	Columbia	mixed spring	24	muscle w/skin	9.0	38	22.0	NR	17

		River	Chinook							
	Oregon	Columbia River	fall Chinook	4	whole body	9.4	15	NR	2.30	15
	Oregon	Columbia River	Clackamas River	3	muscle w/skin	8.8	13	NR	1.80	15
	Oregon	Columbia River	Clackamas River	3	muscle w/o skin	6.1	10	NR	1.50	15
		Oregon Chinook salmon average				8.1	22	24.4	3.53	
	California	Sacramento /San Joaquin	unknown	29	whole body	NR	14	33.6	2.56	10
	Chinook salmon Average					5.6	29	15.7	6.22	
Sockeye salmon	Alaska	Unknown	Alaska	2	muscle w/o skin	NR	3.6	NR	0.21	4
	Alaska	Aleutian Islands	unknown	13	muscle w/o skin	5.8	130	6.9	NR	3
	Alaska	Kodiak	unknown	3	muscle w/skin	NR	5.0	NR	0.10	5*, 6*
	Alaska	Gulf of Alaska/ Berring Sea	unknown	24	muscle w/o skin	8.2	13	12.0	0.22	20
	Alaska	Gulf of Alaska/ Berring Sea	Copper River	97	muscle w/o skin	5.5	37	12.2	NR	18**
	Alaska	SE Alaska	unknown	3	muscle w/skin	NR	13.3	NR	0.10	5*, 6*
		Alaskan sockeye salmon average				6.5	14.4#	10.4	0.16	
	British Columbia	Unknown	unknown	3	muscle w/skin	NR	8.0	NR	0.10	5*, 6*
	British Columbia	Fraser River	Early Stuart	3	Soma	16	13	NR	NR	7**
	British Columbia	Fraser River	Early Stuart	5	muscle w/o skin	4.0	3.9	NR	NR	7**
	British	Fraser River	Early Stuart	6	muscle w/o	5.0	6.9	NR	NR	7**

	Columbia				skin					
	British Columbia	Fraser River	Adams	5	muscle w/o skin	8.8	7.7	6.6	NR	16**
	British Columbia	Fraser River	Weaver Creek	3	muscle w/o skin	1.4	6.8	NR	NR	7**
	British Columbia	Fraser River	Weaver Creek	2	muscle w/o skin	1.1	3.6	NR	NR	7**
	British Columbia	Fraser River	Weaver Creek	2	muscle w/o skin	1.5	5.3	NR	NR	7**
	British Columbia	Fraser River	Weaver Creek	1	muscle w/o skin	1.1	4.0	NR	NR	7**
	British Columbia	Fraser River	Weaver	8	muscle w/o skin	3.9	6.8	5.4	NR	16**
	British Columbia	West Coast VI	Great Central Lk.	6	Muscle	6.1	1.7	NR	NR	7**
	British Columbia	West Coast VI	Great Central Lk.	3	Muscle	6.6	1.6	NR	NR	2**
	British Columbia	West Coast VI	Great Central Lk.	2	Muscle	1.0	1.5	NR	NR	2**
	British Columbia	West Coast VI	Great Central Lk.	3	Muscle	1.0	2.4	NR	NR	2**
	British Columbian sockeye salmon Average					4.4	5.2	6.00	0.10	
	Sockeye salmon Average					4.8	7.6 #	8.6	0.15	
Steel-head	Oregon	Columbia River		21	muscle w/skin	6.0	34	21.0	NR	17
Coho Salmon	Alaska	Unknown	unknown	2	muscle w/o skin	NR	1.6	NR	0.32	4
	Alaska	Kodiak	unknown	3	muscle w/skin	NR	4.0	NR	0.10	5*, 6*
	Alaska	seak/goa	unknown	14	muscle w/o skin	2.9	2.0	1.5	0.19	20
	Alaska	SE Alaska	unknown	3	muscle w/skin	NR	4.0	NR	0.10	5*, 6*
	Alaskan coho salmon Average					2.9	2.9	1.5	0.18	

	British Columbia	Unknown	unknown	3	muscle w/skin	NR	6.0	NR	0.30	5*, 6*
	Washington	Puget Sound	unknown	32	muscle w/o skin	3.1	35	NR	NR	9
	Washington	Puget Sound	PS mixed	125	muscle w/o skin	3.1	27	NR	NR	9
	Washington	Puget Sound	PS mixed	266	muscle w/o skin	3.3	NR	11.7	NR	19
	Washington coho salmon Average					3.2	31	11.7	NR	
	Oregon	Columbia River	Umatilla River	3	muscle w/skin	2.5	35	41.0	NR	17
	Coho salmon Average					3.0	14	18.1	0.20	
Pink salmon	Alaska	Kodiak	unknown	3	muscle w/skin	NR	3.0	NR	0.10	5*, 6*
	Alaska	northern Alaska	unknown	7	Canned	6.3	2.6	1.8	NR	21
	Alaska	SE Alaska/GOA	unknown	12	muscle w/o skin	3.5	1.3	0.6	0.22	20
	Alaska	SE Alaska	unknown	3	muscle w/skin	NR	2.0	NR	0.10	5*, 6*
	Alaskan pink salmon Average					4.9	2.2	1.2	0.14	
	British Columbia	Unknown	unknown	3	muscle w/skin	NR	3.0	NR	0.30	5*, 6*
	Pink salmon Average					4.9	2.4	1.2	0.18	
Chum salmon	Alaska	Kodiak	unknown	3	muscle w/skin	NR	2.0	NR	0.10	5*, 6*
	Alaska	SE Alaska	unknown	3	muscle w/skin	NR	3.0	NR	0.10	5*, 6*
	Alaska	Bering Sea	unknown	18	muscle w/o skin	4.8	3.2	1.9	0.16	20
	Alaskan chum salmon Average					4.8	2.7	1.9	0.12	
	British Columbia	Unknown	unknown	3	muscle w/skin	NR	2.0	NR	0.20	5*, 6*
	Chum salmon Average					4.8	2.6	1.9	0.14	

(1) Cullon et al. 2009, (2) Debruyn et al. 2004, (3) Hardell et al. 2010, (4) Hayward et al. 2007, (5) Hites et al. 2004a, (6) Hites et al.

2004b,
(7) Kelly et al. 2007, (8) Missildine et al. 2005, (9) O'Neill et al. 1998, (10) O'Neill et al. 2006, (11) O'Neill and West 2009,
(12) Rice and Moles 2006, (13) Shaw et al. 2008, (14) Shaw et al. 2006, (15) Stone 2006, (16) Veldhoen et al. 2010,
(17) US EPA 2002, (18) Ewald et al. 1998, (19) West et al. 2001, (20) ADEC 2011, (21) O'Hara et al. 2005
NR is Not Reported
* estimated values from figure
** estimated value from reported lipid weight
#excluded value as an outlier

Extinction Risk. In conjunction with the 2004 status review, NMFS conducted a population viability analysis (PVA) for Southern Resident killer whales (Krahn et al. 2004). Demographic information from the 1970s to fairly recently (1974-2003, 1990-2003, and 1994-2003) were considered to estimate extinction and quasi-extinction risk. NMFS defined "quasi-extinction" as the stage at which 10 or fewer males or females remained a threshold from which the population was not expected to recover.

The model evaluated a range in Southern Resident survival rates, based on variability in mean survival rates documented from past time intervals (highest, intermediate, and lowest survival). The model used a single fecundity rate for all simulations. The study considered seven values of carrying capacity for the population ranging from 100 to 400 whales, three levels of catastrophic event (e.g., oil spills and disease outbreaks) frequency ranging from none to twice per century, and three levels of catastrophic event magnitude in which 0, 10, or 20 percent of the animals died per event.

The analysis indicated that the Southern Resident killer whales have a range of extinction risk from 0.1 to 18.7 percent in 100 years and 1.9 to 94.2 percent in 300 years, and a range of quasi-extinction risk from 1 to 66.5 percent in 100 years and 3.6 to 98.3 percent in 300 years (Table 5). The population is generally at greater risk of extinction as survival rate decreases and over a longer time horizon (300 years) than over a shorter time horizon (100 years) (as would be expected with long-lived mammals). There is a greater extinction risk associated with increased probability and magnitude of catastrophic events. The NWFSC continue to evaluate mortality rates and reproduction, and will complete work on a PVA similar to the analysis summarized above. Until these updated analyses are completed, the Krahn et al. 2004 analysis represents the best available science on extinction risk of Southern Resident killer whales.

Table 5. Range of extinction and quasi-extinction risk for Southern Resident killer whales in 100 and 300 years, assuming a range in survival rates (depicted by time period), a constant rate of fecundity, between 100 and 400 whales, and a range catastrophic probabilities and magnitudes (Krahn et al. 2004).

Time Period	Extinction Risk (%)		Quasi-Extinction Risk (%)	
	100 yrs	300 yrs	100 yrs	300 yrs
Highest survival	0.1 – 2.8	1.9 – 42.4	1.0 – 14.6	3.6 – 67.7
Intermediate survival	0.2 – 5.2	14.4 – 65.6	6.1 – 29.8	21.4 – 85.3
Lowest survival	5.6 – 18.7	68.2 – 94.2	39.4 – 66.5	76.1 – 98.3

2.2.2 Status of Critical Habitat

Puget Sound Chinook Salmon Critical Habitat

At this time, CH is designated for only one species of fish affected by the proposed action, Puget Sound Chinook salmon. The NMFS designated CH for Puget Sound Chinook salmon on September 2, 2005 (70 FR 52630).

The NMFS reviews the status of designated CH affected by the proposed action by examining the condition and trends of the PCEs throughout the designated area. The PCEs are physical features essential to the conservation of the ESU (for example, spawning gravels, good water quality and appropriate water quantity, accessible side channels, sufficient forage species), because these features enable spawning, rearing, migration, and foraging behaviors essential for survival and recovery. Specific types of sites, and the features associated with the PCEs for salmonids, include:

- Freshwater spawning sites with water quantity and quality conditions and substrate supporting spawning, incubation and larval development.

- Freshwater rearing sites with: (i) water quantity and floodplain connectivity to form and maintain physical habitat conditions and support juvenile growth and mobility; (ii) water quality and forage supporting juvenile development; and (iii) natural cover such as shade, submerged and overhanging large wood, log jams and beaver dams, aquatic vegetation, large rocks and boulders, side channels, and undercut banks.

- Freshwater migration corridors free of obstruction and excessive predation with water quantity and quality conditions and natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, side channels, and undercut banks supporting juvenile and adult mobility and survival.

- Estuarine areas free of obstruction and excessive predation with: (i) water quality, water quantity, and salinity conditions supporting juvenile and adult physiological transitions between fresh- and saltwater; (ii) natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, side channels; and (iii) juvenile and adult forage, including aquatic invertebrates and fishes, supporting growth and maturation.

- Nearshore marine areas free of obstruction and excessive predation with: (i) water quality and quantity conditions and forage, including aquatic invertebrates and fishes, supporting growth and maturation; and (ii) natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, and side channels.

- Offshore marine areas with water quality conditions and forage, including aquatic invertebrates and fishes, supporting growth and maturation.

All PCEs of freshwater (spawning, rearing, and migratory life stages), estuarine and nearshore marine (juvenile development and growth) CH have been degraded by a variety of human activities, throughout the Puget Sound region, some more than others. Extensive areas of Chinook salmon spawning and rearing habitat have declined because of: (1) degraded floodplain and in-river channel structure, complexity, and connectivity; (2) degraded estuarine conditions and loss of estuarine habitat; (3) riparian area degradation and loss of in-river large woody debris; (4) excessive sediment in spawning gravels; and (5) degraded water quality and temperature (NMFS 2007a).

Within freshwater environments, watershed development and associated urbanization throughout Puget Sound, Hood Canal, and Strait of Juan de Fuca regions have increased sedimentation, raised water temperatures, decreased large woody material recruitment, decreased gravel recruitment, reduced river pools and spawning areas, and dredged and filled estuarine rearing areas (Bishop & Morgan 1996). Forestry practices, farming and urbanization have blocked or degraded freshwater habitat (Meyers et al. 1998). Large areas of lower river meanders (formerly mixing zones between fresh and salt water) have been channelized and diked for flood control and to protect agricultural, industrial and residential development. In general, forest practices impacted upper tributaries, and agriculture or urbanization impacted lower tributaries and Main stem Rivers. The WDFW and Western Washington Treaty Indian Tribes (WWTIT) cited diking for flood control, draining and filling of freshwater and estuarine wetlands, and sedimentation due to forest practices and urban development as problems throughout the ESU (WDFW & WWTIT 1994a; WDFW & WWTIT 1994b; WDFW & WWTIT 1994c; WDFW & WWTIT 1994d).

Estuarine areas are becoming increasingly degraded (Bishop & Morgan 1996). The sub-estuaries of Puget Sound—the major river deltas—have suffered a collective 80 percent loss of tidal marsh habitats in the past 150 years (Dean et al. 2001). Continued development and modification of the Puget Sound shoreline contributes to a cumulative degradation or loss of near shore and estuarine habitat, identified by Thom et al., (1994) as the most harmful loss to juvenile salmonids. As described above, the three most common life history types of juvenile Chinook salmon, fry migrants, delta migrants, and parr migrants, all heavily rely on the marine nearshore. The development of shorelines includes the introduction of obstructions in the nearshore, and overwater structures, which impede juvenile migration.

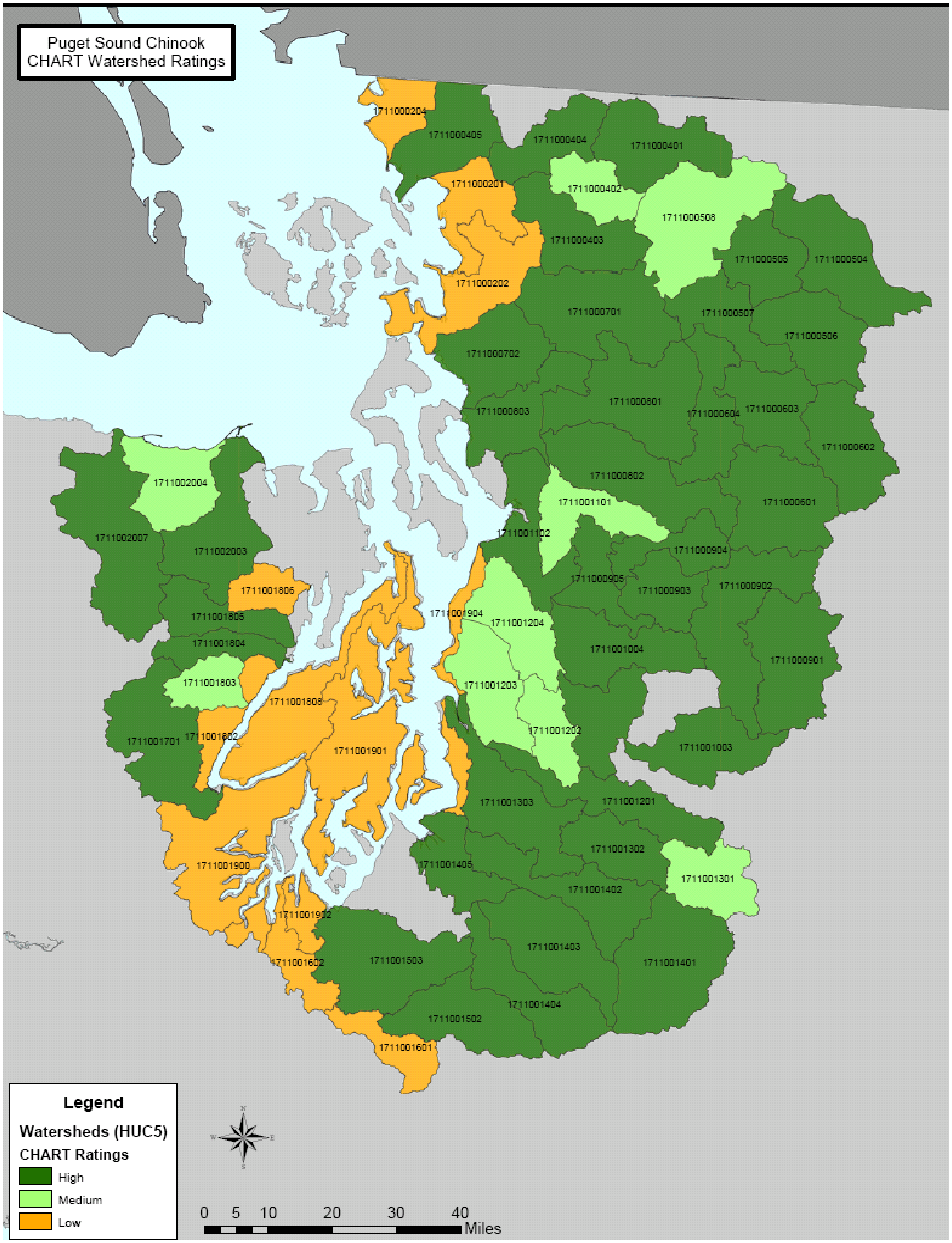
Of nearshore and marine habitat, at least 33 percent of Puget Sound shorelines (813 miles) have been modified with bulkheads or other armoring, and one-half of this is associated with single-family residences (Puget Sound Water Quality Action Team 2002). The Washington State Nearshore Inventory (Nearshore Habitat Program 2001) reports thousands of overwater structures in Puget Sound in the late 1990s to 2000, including 3,500 piers and docks; 29,000 small boat slips; and 700 large ship slips. Each is a source of structure and shade which can support predator fish, interfere with juvenile salmonid migration, diminish aquatic food supply, and is a potential source of water pollution and harassment from boating uses.

The NMFS ranked the conservation value of watersheds (Fifth Field Hydrologic Unit Codes [HUCs]) in the designated range of Puget Sound Chinook salmon critical habitat. Conservation value rankings considered the presence of the primary constituent elements of critical habitat in the HUC, the salmonid life history stages expressed in the HUC, and the relative importance of component populations relative to overall ESU viability, they support[2]. Conservation rankings

[2] The conservation value of a site depends upon "(1) the importance of the populations associated with a site to the ESU [or DPS] conservation, and (2) the contribution of that site to the conservation of the population through demonstrated or potential productivity of the area" (NMFS 2005).

are high, medium, or low (Figure 6). To determine the conservation value of each watershed to ESU viability, the CHART (Critical Habitat Analytical Review Team) evaluated the quantity and quality of habitat features (for example, spawning gravels, wood and water condition, side channels), the relationship of the area compared to other areas within the ESU, and the significance to the ESU of the population occupying that area. Thus, even a location that has poor quality of habitat could be ranked at high conservation value if that location was essential due to factors such as limited availability (e.g., one of a very few spawning areas), the unique contribution of the population it served (e.g., a population at the extreme end of geographic distribution), or other important role (e.g., obligate area for migration to upstream spawning areas).

The Puget Sound Chinook salmon ESU has 61 freshwater and 19 marine areas within its range. Of the freshwater watersheds, 41 are rated with a high conservation value, 12 have a low conservation value, and eight received a medium rating. Of the marine areas, all 19 are ranked with high conservation value.


Puget Sound Chinook
CHART Watershed Ratings
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1711000405
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1711000403
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Legend
Watersheds (HUC5)
CHART Ratings
High
Medium
Low
0 5 10 20 30 40
Miles

Southern Resident Killer Whales Critical Habitat

The final designation of critical habitat for the Southern Resident killer whale DPS was published on November 29, 2006 (71 FR 69054). Critical habitat consists of three specific areas: (1) the Summer Core Area in Haro Strait and waters around the San Juan Islands; (2) Puget Sound; and (3) the Strait of Juan de Fuca. These areas comprise approximately 2,560 square miles of marine habitat. Based on the natural history of the Southern Residents and their habitat needs, NMFS identified the following physical or biological features essential to conservation: (1) Water quality to support growth and development; (2) Prey species of sufficient quantity, quality and availability to support individual growth, reproduction and development, as well as overall population growth; and (3) Passage conditions to allow for migration, resting, and foraging.

Water Quality. Water quality in Puget Sound, in general, is degraded as described in the Puget Sound Partnership Recommendations and subsequent Action Agenda (Puget Sound Partnership 2006, 2008). For example, toxins in Puget Sound persist and build up in marine organisms including Southern Residents and their prey resources, despite bans in the 1970s of some harmful substances and cleanup efforts. The primary concern for direct effects on whales from water quality is oil spills, although oil spills can also have long-lasting impacts on other habitat features. The Environmental Protection Agency and U.S. Coast Guard oversee the Oil Pollution Prevention regulations promulgated under the authority of the Federal Water Pollution Control Act. There is a Northwest Area Contingency Plan, developed by the Northwest Area Committee, which serves as the primary guidance document for oil spill response in Washington and Oregon. In 2007, the Washington State Department of Ecology published a new Spill Prevention, Preparedness, and Response Program Annual Report describing recent accomplishments and declining trends in spill incidents per transit (WDOE 2007).

Prey Quantity, Quality, and Availability. As discussed above under Limiting Factors and Threats, most wild salmon stocks throughout the Northwest are at fractions of their historic levels. Beginning in the early 1990s, 28 ESUs and DPSs of salmon and steelhead in Washington, Oregon, Idaho, and California were listed as threatened or endangered under the ESA. Historically, overfishing, habitat losses, and hatchery practices were major causes of decline. Poor ocean conditions over the past two decades have reduced populations already weakened by the degradation and loss of freshwater and estuary habitat, fishing, hydropower system management, and hatchery practices. While wild salmon stocks have declined in many areas, hatchery production has been generally strong. Total Chinook abundances coastwide increased significantly from the mid-1990s to the early 2000s, but have declined in the last several years (PFMC 2008).

Contaminants and pollution also affect the quality of Southern Resident killer whale prey in Puget Sound. Contaminants enter marine waters and sediment from numerous sources, but are typically concentrated near areas of high human population and industrialization. Once in the environment these substances proceed up the food chain, accumulating in long-lived top predators like Southern Resident killer whales. Chemical contamination of prey is a potential threat to Southern Resident killer whale critical habitat, despite the enactment of modern pollution controls in recent decades, which were successful in reducing, but not eliminating, the presence of many contaminants in the environment. The size of Chinook salmon is also an important aspect of prey quality (i.e., Southern Residents primarily consume large Chinook, as discussed above), and any reduction in Chinook salmon size can affect the prey feature and the conservation value of their critical habitat. In addition, vessels and sound may reduce the effective zone of echolocation and reduce availability of fish for the whales in their critical habitat (Holt 2008).

Passage. Southern Residents killer whales are highly mobile and use a variety of areas for foraging and other activities, as well as for traveling between these areas. Human activities can interfere with movements of the whales and impact their passage. In particular, vessels may present obstacles to whale passage, causing the whales to swim further and change direction more often, which can increase energy expenditure for whales and impact foraging behavior (review in NMFS 2011c).

2.3 Environmental Baseline

The "environmental baseline" includes the past and present impacts of all Federal, state, or private actions and other human activities in the action area, the anticipated impacts of all proposed Federal projects in the action area that have already undergone formal or early section 7 consultation, and the impact of state or private actions which are contemporaneous with the consultation in process (50 CFR 402.02).

Throughout Puget Sound, as in the action area, the marine environment, sloughs, marshes, eelgrass and kelp beds have been destroyed and degraded by pervasive vegetation removal and construction along stream banks and shorelines. The structural diversity provided by the complex interaction of light, water, soil, vegetation and nutrient cycles that salmon evolved to, have an increasing the frequency of human disturbance, altering the magnitude of disruption, and affecting the ability of the environment to respond. Of the nearly 66,000 acreas of tidal marsh and wetlands bordering Puget Sound in 1885, approximately 100 years later, only 13,500 acres of intertidal marine or vegetated habitat is estimated to remain in the Puget Sound basin. This represents a decline of 80 percent across the region due to agricultural and urban modification of the lowland landscape (NMFS/Chum BRT 1997).

In addition to the high-intensity industrial and urban development at the major river mouths in Puget Sound, intertidal and nearshore habitats throughout the Sound have been modified by shoreline armoring (e.g. construction of rock, concrete, and timber bulkheads or retaining walls). Approximately 33 percent of Puget Sound Shorelines have been modified with bulkheads or other armoring. These modifications have a cumulative environmental impact that results in loss of riparian vegetation, obstruction of sediment movement along the shoreline, interference with wave action, and burial of upper beach areas. Although upper beach areas are not utilized directly by salmon, they are egg-laying grounds for species of smaller forage fish that salmon depend on.

These factors combined with the effects of upland forestry, urban construction, and rural agricultural practices have decreased sediment quality and function as an element of salmonid habitat. According to the NPDES Fact Sheet (page 24 of 47), marine sediment monitoring was conducted in 1995 at 14 sites near the outfall and at one reference location in Carr Inlet in the action area. All samples were taken from the upper two centimeters of bottom sediment layer. They then underwent chemical analysis and biological tests were performed. The chemical concentrations met the Washington State Department of Ecology, Sediment Management Standards (SMS) criteria for SMS chemicals except bis(2-ethylhexyl)phthalate (a common wastewater chemical). Fourteen of the 15 samples had concentrations above the SMS criteria, including the sample from the reference site.

Furthermore, the mix of oil, grease, pesticides and other pollutants carried by stormwater runoff alters the chemical processes of urban streams and creates dramatic shifts in their flow patterns (Shared Strategy 2007). Recent studies by NMFS and Seattle Public Utilities have also documented high rates of outright mortality to adult salmon still full of eggs and sperm, even in a creek where habitat had been restored (McCarthy et al. 2008). While the restoration of these urban creeks is essential to allowing greater numbers to spawn, the studies suggest that the control of polluted runoff from urban streets, lawns and parks and restoration of chemical balance is imperative to fish productivity and population restoration in urban streams (Spromberg & Scholtz 2011).

Wastewater treatment plants contribute additional metals and contaminants such as ammonia, chloride, aluminum, boron, iron, manganese, oil/grease, PCBs and other toxic substances. There are nearly 1000 municipal and industrial wastewater discharges into the Puget Sound Basin that are permitted by the Washington Department of Ecology. Of those, 180 permit holders had specific permission to discharge metals, including mercury and copper. Over 1 million pounds of chemicals were discharged to Puget Sound in 2000 by the 20 industrial facilities that reported their releases to the Environmental Protection Agency.

Flushing Rates

The WDOE is in the process of evaluating levels of dissolved oxygen in South Puget Sound as portions of that area do not meet Washington State water quality standards for parameter (Roberts et al. 2009). A three-dimensional model has been developed that simulates tides, water velocity, temperature, and salinity to describe water circulation in South and Central Puget Sound. It was used to determine the action area depicted in Figure 2. For accuracy it was calibrated with parameters that included water surface elevations, tidal constituents, surface temperature and salinity spatial patterns, temperature and salinity profiles, and current velocities. The model showed that in the vicinity of Ketron Island where the WWTP outfall is located, currents circulate around the island in a clockwise direction some of which flows west into Balch Passage and some going south through the Nisqually reach.

A dye tracer was used by WDOE as an interim indicator of areas influenced by rivers and point source discharges. This encompassed all river inflows and wastewater treatment facilities with flows greater than one million gallons per day, including the Fort Lewis WWTP outfall. The model was used to evaluate how long dye released into South or Central Puget Sound would continue to build up to a pseudo-steady-state condition. While flushing times for South Sound are on the order of a month, model simulations indicated that the dye continues to build up for several months (Roberts et al. 2009).

Albertson et al. (2001), determined residence times in various water bodies of the Southern Puget Sound Basin based on volume and mass transport modeling. Residence time in the Southern Basin was calculated at 124 days. Puget Sound residence times for pollutants are extended because of flow circulation conditions that carry discharged contaminants further into the sound rather than out of it. This is particularly true for suspended solids that have the tendency to settle out when given sufficient residence time, as is found in the side channels of the Puget Sound (LaLiberte & Ewing 2006).

This condition is causing water quality and sediment conditions in Puget Sound to worsen (LaLiberte & Ewing 2006). It also indicates the fate of persistent bioaccumulating toxins is to remain in the Sound and thereby find their way into organisms. The greater the amount of time the mixed effluent spends in Puget Sound, the more opportunity pollutants have to interact with sound sediments, plants and animals. The problem is particularly acute for persistent toxic chemicals like heavy metals and organic compounds, which stay in the sound and do not break down, break down very slowly, or convert into other harmful chemicals. The extended residence times cause toxic pollutants to be persistently recycled through water, sediments and biota (LaLiberte & Ewing 2006).

Carr/Nisqually Subbasin Habitat Conditions

The Carr/Nisqually subbasin, in which the action area is located, is part of South Puget Sound as defined by Burns (1985). The subbasin includes the marine waters and related nearshore located south of the Tacoma Narrows Bridge forming the southern end of the larger Puget Sound fjord estuary complex. Geographically South Puget Sound is separated from Central Puget Sound by a narrow, shallow sill associated with the Tacoma Narrows. The constriction at the Tacoma Narrows results in extreme tidal ranges, up to 18 feet, nearly twice the tidal range as observed in the Strait of Juan de Fuca. The total surface area of South Puget Sound's marine waters is approximately 394 square kilometers.

More than 50 percent of South Puget Sound is less than 36.6 meters deep and only a very small percentage is deeper than 100 meters (Burns 1985). South Puget Sound is divided into numerous shallow, blind-end inlets, causing poor circulation. Consequently, water does not mix or dilute nutrient inputs to the same degree as the deeper, more tidally mixed areas elsewhere in the Puget Sound basin. The shallow nature of South Puget Sound provides a wider nearshore fringe than exists in Central and North Puget Sound and, along with the slow flushing time, makes many of the bays and inlets more productive then the rest of Puget Sound (WDOE 2006).

The Carr/Nisqually subbasin has about 156 miles of shoreline, of which 44 percent (68 miles) are armored. Redman et al. (2005) reported 1,588 overwater structures in the basin, of which 177 are ramps, 346 piers and docks, 1,058 small slips, and 7 large slips at an average of about 10 overwater structures per mile. This concentration is relative low concentration of overwater structures compared to the 48 overwater structures per mile in all of South and Central Puget Sound combined, see Table 2 below.

Eelgrass is present in 34 percent of the shoreline in the subbasin, about 53 miles. Eelgrass abundance declines toward the southern end of South Puget Sound, where it is subject to desiccation, presumably because of the extreme tidal range. At extreme high tides, light does not penetrate the water to a sufficient intensity to sustain eelgrass growth. Non-floating kelp only occurs along 6 linear miles, or 4 percent of the shoreline (Redman et al. 2005).

Table 6: Habitat Statistics for South Puget Sound Relative to Central Puget Sound (Redmond et al. 2005)

Basin	Shore-line, miles	Armoring, miles	Percent Armored	Near-shore (acres)	Over Water Structure (OWS)	OWS / mile	Eelgrass (shore-line miles)	NF kelp (shore-line miles)
South Puget Sound	293	109	37.20	34496	2,626	9	10	93
Carr Nisqually	156	68	43.59	16448	1588	10	53	6
Port Madison /Sinclair Inlet	96	56	58.33	13376	22383	233	15	17
Central Puget Sound	308	179	58.12	33856	10448	34	154	71
Sum	853	412	48.30	98176	37,045	43	232	187

The Carr/Nisqually subbasin also contains a high concentration of pocket estuaries compared to other parts of Puget Sound. The many pocket estuaries are distributed relatively uniformly throughout the subbasin. (Redman et al. 2005) These pocket estuaries are important for in- and out of- basin rearing Chinook salmon.

Timber in riparian and lowland areas has been systemically removed starting in the 1870's, reducing shade, cover, and the food supply for salmon in both fresh and saltwater environments, as fewer large trees and root wads enter aquatic systems, including the saltwater shore zones of Puget Sound. Consequently, protection for rearing and migrating salmon was diminished. In addition to these long term declines in habitat features of the action area, the entire shoreline of the action area has railroad tracks along its length, with extensive armoring of the bank to support the rail line. Only a few short segments of the bank and shore are left in its natural condition.

The Nisqually River estuary, located on the southern edge of the action area, is currently the best estuarine salmon habitat in the region following a yearlong restoration of the delta that greatly increased the available estuarine habitat in November 2009. The Brown Farm Dike was removed to inundate 761 acres of the Nisqually National Wildlife Refuge in November 2009, along with 141 acres of wetlands previously restored by the Nisqually Indian Tribe. The Nisqually Delta now represents the largest tidal marsh restoration project in the Pacific Northwest and the Nisqually River now has the largest undeveloped delta in Puget Sound.

According to the Washington State Department of Fish and Wildlife's Salmon-Scape database, surf smelt spawning habitat has been documented within 500 feet, south of the outfall. Within 2000 feet to the north there is a boat launch and an independent, unnamed tributary to Puget Sound. According to WDOE's Coastal Atlas Maps there is patchy or continuous presence of kelp throughout much of the action area. This includes the immediate vicinity of the WWTP outfall. There is a lack of specific habitat information for deeper portions of the nearshore with regard to the mixing zone and surrounding area.

2.3.1 Species Status in the Action Area

The Nisqually River is located a little more than three miles south of the action area. While the Nisqually River is the only major river system entering the Carr/Nisqually subbasin, use of the action area is predominantly by three Chinook salmon populations (Nisqually, White and Puyallup) because the Nisqually, White and Puyallup Rivers are in closest proximity to the action area. However, six or more populations of Puget Sound Chinook salmon and one population of Puget Sound steelhead have been documented in the action area. Use has been noted among Chinook salmon from the Duwamish/Green River, Snohomish River (includes the Snoqualmie and Skykomish Rivers) and the Skagit River (includes Lower Skagit, Upper Skagit, Cascade, Lower Sauk, Upper Sauk and Suiattle Rivers, (Duffy 2003). The main steelhead population making use of the action area would be of Nisqually River origin. Redman et al. (2005) also reported that the Carr/Nisqually subbasin supports rearing Chinook salmon from six independent populations, the Cedar/Lake Washington, Green (spring and fall Chinook salmon), Puyallup, White and Nisqually. These six populations can be grouped together as the South Sound Group by Diversity and Risk (Figure 5). Redman et al. (2005) thought that parr migrants moving south out of the Central Puget Sound subbasin utilize and greatly depend on the shoreline habitats within the Carr-Nisqually subbasin.[3]

Chinook salmon fry migrants, delta migrants, and parr migrants from the Nisqually natal population utilize the Nisqually/Carr subbasin nearshore for feeding, refuge, physiological transition and as a migratory corridor. The Puyallup River (17 miles North) is the other closest spawning tributary to the Solo Point outfall. Because of proximity, NMFS considers Chinook salmon and steelhead from these two populations to be the most predominant found in the action area.

Since Vashon and Maury Island have no Chinook salmon-bearing streams, the presence of Chinook salmon at these sites means that salmon have been crossing an open, deep water channel away from the protection of the nearshore environment. The minimum straight-line traveled from release location to sample location for hatchery Chinook salmon is impressive. The distance crossed ranged from 12 km to 267 km. While traveling these long distances in deep open water, Chinook salmon from North Puget Sound stocks also moved south and southeast.

Some juvenile Chinook salmon do not simply leave their natal stream and migrate north and out of Puget Sound, which has been shown by other research. Most recently, Duffy et al. (2005)

[3] This is supported by other research. Brennan et al. (2004) reports for nearshore seines, no significant difference between the catch of Chinook salmon at island sites compared to mainland sites.

reported that 40 percent of the CWT Chinook salmon recaptured at nearshore sampling sites between the Nisqually River and the Tacoma Narrows were from hatcheries in the central basin of Puget Sound.

Nisqually River Chinook Salmon

The Nisqually Chinook salmon population has been impacted by hatchery practices, habitat degradation, and high harvest rates. As a result, native Nisqually Chinook salmon have been extirpated, and the current production consists primarily of hatchery releases (between 1999-2008 escapement averaged 68 percent hatchery fish) with some natural spawning in the main stem and lower reaches of major tributaries. At 0.92, the median growth rate (lambda), between 1990 and 2005, for return (i.e., recruits/spawners) shows a substantial declining trend. Median growth rate for escapement (i.e., spawners/spawners), for the same time frame, was slightly increasing at 1.01. The geometric mean for escapement (spawners) from 1999-2009 of natural fish (includes naturally spawning hatchery stock) was 1,549 (NMFS 2010a). The draft Puget Sound Chinook Salmon Population Recovery Approach has assigned this population a Tier 1 ranking (NMFS 2010b). "Tier 1" populations are the primary populations that are most important for preservation, restoration, and ESU recovery.

Duffy (2003) found that juvenile Chinook salmon occur in nearshore Puget Sound waters for at least six months of the year (April through September). Peak occurrence spanned a three month time frame from May through July. Sampling locations occurred from Solo Point north towards the Tacoma Narrows. Juvenile Chinook salmon are known to spend 6-16+ weeks in Puget Sound and Hood Canal estuaries with individuals remaining for 1-7 weeks (Simenstad et al. 1982). Some resident Chinook salmon remain in Puget Sound until maturity (Simenstad et al. 1982). Juvenile fish from the Nisqually population will use the action area (April through September) and possibly contribute to the resident population in South Puget Sound. Adults likely pass through relatively quickly during the months of July and August on their way to spawn in the river.

Puyallup and White River Chinook Salmon

The Puyallup River basin supports two populations of Chinook salmon, early returning White River Chinook salmon spawning in the upper and lower White River, and late returning Chinook salmon spawning in the Carbon River, Puyallup River, and associated tributaries. There are also some late returning Chinook salmon that spawn in the lower White River that have not yet been assigned to a specific population.

The geometric mean of natural spawner abundance between 1999 and 2009 was 987 for the White and 969 for the Puyallup. Median growth rates between 1990 and 2005 are increasing for the White River fish, 1.13 for return and 1.12 for escapement, and declining for the Puyallup River fish, 0.88 for return and 0.91 for escapement (NMFS 2010a). The draft Puget Sound Chinook Salmon Population Recovery Approach has assigned the White River population a Tier 1 ranking. Tier 1 populations are the primary populations that are most important for preservation, restoration, and ESU recovery. The Puyallup River population was assigned a Tier 3 ranking (NMFS 2010b).

Both Chinook salmon populations have been adversely affected by hydroelectric dams, impassable culverts, lack of estuarine and nearshore habitat, diversion of flows, impaired water quality, and impaired riparian functions. The Mud Mountain Dam and White River Hydroelectric Project have eliminated 9.6 miles of main stem spawning and rearing habitat. Returning adult salmon are trapped at the diversion dam and trucked upstream of the Mud Mountain Dam impoundment where they are released back into the White River at RM 33.9. About 70 percent of the known culverts within the Puyallup river watershed in 1999 acted as partial barriers to salmon migration upstream and downstream; about 40 percent were determined to be complete barriers.

Out of more than 5,900 acres of estuary habitats that historically existed at the head of Commencement Bay, only about 200 acres remain due to dredging, filling and activities associated with development. The substantial loss of estuary habitat support for the Chinook salmon populations has reduced capacity, productivity, and diversity. Contaminated sediments which have further limited the nearshore/estuarine habitat have resulted in additional reductions in Chinook salmon productivity. Diversion of flows from the 24 mile bypass reach of the lower White River and the ten mile reach of the Puyallup River between the Electron Powerhouse and the dam have reduced spawning and rearing habitat and disrupted the use of the river as a migratory corridor. Periodic manipulations of flows associated with operations at both facilities are believed to result in recurrent fish stranding and kills.

Juvenile fish from the Puyallup and White River populations will use the action area (April through September) and possibly contribute to the resident population in South Puget Sound. A few adults are likely to quickly pass through the action area during the months of May through August on their way upstream to spawn in these respective rivers.

Green/Duwamish River Chinook Salmon

The Green/Duwamish Chinook population is an integrated wild-hatchery population with a major role played by hatchery fish. There are several hatcheries operated by the Muckleshoot Tribe and Washington State Department of Fish and Wildlife. Chinook salmon in this basin return to spawn in the summer and fall. Some of the hatchery fish spawn, as the wild-origin fish do, in the main stem reaches of the Middle Green River, in Soos Creek and in Newaukum Creek.

The geometric mean of natural spawner abundance between 1999 and 2009 was 3,615 adults. Median growth rates of the population between 1990 and 2005 are increasing, 1.01 for return and 1.04 for escapement (NMFS 2010). The draft Puget Sound Chinook Salmon Population Recovery Approach has assigned the Green/Duwamish River population with the only Tier 2 ranking in the Central and South Sound biogeographical region. This will help ensure that at least one population in the region is recovered at a sufficient pace to allow for its potential inclusion as a "Tier 1" population if needed for recovery (NMFS 2010b).

Juvenile fish from the Green/Duwamish River population will make use of the action area (April through September) and could possibly contribute to the resident population in South Puget

Sound. Few adults will likely pass through the action area during the months of July and August on their way to spawn in the river.

Snoqualmie and Skykomish (Snohomish) River Chinook Salmon

Since the late 1970s, the Skykomish population has experienced a steep decline in total number of fish. Between 1999 and 2008, the Skykomish population has averaged about 2,578 natural origin fish that return to the river to spawn. The Snoqualmie has averaged approximately 1,731 between 1999 and 2009 g (NMFS 2010a). Together this means that the populations are at approximately 5 percent and 5.7 percent of their historic numbers respectively. These numbers do not include hatchery fish that return to the natural spawning ground. The Skykomish run has the highest recovery target for abundance of those set for Puget Sound Chinook populations; the Snoqualmie run has the third highest target. The median growth rate of the Skykomish River population between 1990 and 2005 was stable at 0.99 for return and increasing for escapement at 1.05. For the Snoqualmie River it was 0.99 for return (stable) and 1.03 (increasing) for escapement (NMFS 2010a).

The draft Puget Sound Chinook Salmon Population Recovery Approach has assigned the Skykomish River population a Tier 2 ranking and the Snoqualmie River population was assigned a Tier 3 ranking (NMFS 2010b).

Juvenile fish from this population will make use of the action area (April through September) and could possibly contribute to the resident population in South Puget Sound. Few, if any, adults will likely pass through the action area during the months of July and August on their way to spawn in these rivers.

Upper Cascade, Suiattle, Upper Sauk, Lower Sauk, Upper and Lower Skagit River Chinook Salmon

There are six Chinook populations that exist within the Skagit River Basin. The Upper Cascade, Suiattle, and Upper Sauk populations comprise the Spring Management Unit. The Upper and Lower Skagit and Lower Sauk populations comprise the Fall/Summer Management Unit. The geometric mean of natural spawner abundance between 1999 and 2009 for the Upper Cascade, Suiattle and Upper Sauk Rivers was 425, 317 and 298 respectively. For the Upper and Lower Skagit and Lower Sauk Rivers natural spawner abundance for the same time period was 10,561; 690; and 2,248 respectively. Median growth rates (return and escapement) between 1990 and 2005 Skagit River populations were 1.05 and 1.05 for Upper Cascade (both are increasing); 0.99 and 0.99 for Suiattle (both are stable); 0.95 (declining) and 1.00 (stable) for Upper Sauk; 0.98 (stable) and 1.06 (increasing) for Upper Skagit; 0.97 (stable) and 1.02 (increasing) for Lower Skagit; and 0.97 and 1.00 (both stable) for the Lower Sauk (NMFS 2010a).

The draft Puget Sound Chinook Salmon Population Recovery Approach assigned all of the Skagit River populations with a Tier 1 ranking. Tier 1 populations are the primary populations that are most important for preservation, restoration, and ESU recovery (NMFS 2010b).

Juvenile fish from these populations will use the action area (April through September) and could possibly contribute to the resident population in South Puget Sound. Adults from these populations are not likely to pass through the action area on their way to spawn in these rivers.

Nisqually River Winter-run Steelhead

According to a Washington Department of Fish and Wildlife (2008) report, pre-settlement distribution encompassed anywhere from 171-198 miles of habitat in this river basin. There has been a loss of 7-33 miles, which equals 4-17 percent. The spatial structure is predicted to have been reduced by 43 percent relative to pre-settlement conditions.

The geometric mean of estimated adult escapement of naturally produced steelhead from 2005-2009 is 402 fish (Ford et al. 2010). Trends in escapement and run size have obviously been on the decline. The median short-term population growth rate estimate for the years 1995-2009 is 0.935 (Ford et al. 2010).

The migration pattern of steelhead in Puget Sound is not well understood; it is generally thought that steelhead smolts move quickly offshore (Hartt & Dell 1986). Juvenile steelhead will make some use of the action area during the time period of April to mid-May. Fresh et al., (1979) captured low numbers of juvenile fish using beach seines and tow nets within the action area. Adults would be expected to quickly pass through during the months of December-April on their way to spawn in the river.

Puget Sound/Georgia Basin Yelloweye Rockfish, Canary Rockfish and Bocaccio

Habitat characteristics and depth in this portion of Puget Sound make it unlikely that adult ESA-listed rockfish would use the area encompassed by the mixing zones. They are likely to be found in the action area, however, and as close as a few hundred feet from the mixing zones, within waters deeper than 120 feet. Juvenile and sub-adult canary rockfish or bocaccio that are within the action area would be expected to be found near benthic areas with steep slopes, rock, or kelp beds. There is patchy kelp habitat along some sections of the nearshore (immediately adjacent to the mixing zones) which may be seasonally used by juvenile and sub-adult canary rockfish and bocaccio. It is unlikely that juvenile yelloweye rockfish will occur within kelp habitats of the action area because they don't use the nearshore for rearing. Larval rockfish likely remain in the region they are released within the DPSs (Drake et al. 2010), but may be broadly dispersed from the place of their birth (NMFS 2003). It is expected that larval yelloweye rockfish, canary rockfish, or bocaccio occur within the action area and the mixing zones throughout the year.

Factors Affecting Southern Resident Killer Whales in the Action Area

Toxic Chemicals. Puget Sound is a deep-water fjord with several sills that restrict mixing and inhibit ocean inflow and the outflow of toxic chemicals. Toxic chemicals that enter the basin have longer residence times within the basin resulting in food webs being exposed to higher levels of persistent pollutants. Additionally, many species are known to exhibit a high degree of residency within Puget Sound (e.g., there are several resident populations of fish including Pacific herring and Chinook salmon) resulting in more fish being exposed to more contaminants.

Thus, the Puget Sound ecosystem and food webs are more susceptible to toxic input because of the proximity to urban areas, and the combination of hydrological isolation of the Puget Sound and the biological isolation of resident species (Collier et al. 2006; West et al. 2008; O'Neill and West 2009).

All persistent organic pollutants are contaminants of concern for killer whales, PS Chinook and steelhead, and PS/Georgia Basin yelloweye rockfish, canary rockfish and bocaccio. However, PCBs and DDTs are no longer in use, are declining in the environment, and have protective regulations. In contrast, PBDEs are still in use (although they are in the process of being phased out in WA), and there are no protective regulations. PBDEs are also found in significant and measurable amounts in wastewater effluent (EPA 2010). Thus, in this section, we review PBDE levels in the water and sediment of the action area, and finally PBDE levels in the Southern Residents compared to the Northern Residents.

PBDEs in the Puget Sound Water Column. Recently, a multiphase project was initiated by several agencies (e.g., Washington Department of Ecology and the Puget Sound Partnership) to assess toxic chemical loadings into Puget Sound with the objective to significantly reduce toxics entering Puget Sound fresh and marine waters (see http://www.ecy.wa.gov/programs/wq/pstoxics/index.html for sub-task reports). In the first phase, Crowser et al. (2007) provides initial loading estimates for 17 chemicals of concern (several of which may pose risk to the Southern Residents) from wastewater discharges, and other sources including surface runoff, atmospheric deposition, oil spills, and combined sewer overflows (CSOs). These 17 chemicals of concern included PBDEs, PCBs, dioxin, DDTs, and PAHs (see Table 3 from the Status of the Species section for entire list of chemicals of concern for killer whales). The second phase improved loading estimates from surface runoff, municipal and industrial wastewater discharges, exchange with the ocean, and contaminated sediments. The third and final phase develops their strategy to measure and control the sources of toxics that enter Puget Sound. Additionally, fate and transport models of toxic chemicals were improved upon, and several projects collected and analyzed environmental samples of water, sediment, atmospheric deposition, and various biota.

In the third phase, annual PBDE loadings were estimated from treated wastewater discharge from publicly owned treatment works (POTWs) in Puget Sound (WDOE and Herrera Environmental Consultants, Inc. 2010). The researchers estimated that 7 to 21 kg of total PBDEs are discharged from treated effluent into the Puget Sound each year (a median of 10.6 kg/yr). Other primary sources or pathways of PBDE loadings in the Puget Sound include air deposition (estimated loadings are between 16 and 24 kg/yr with a median of 20.3 kg/yr) and surface runoff (estimated loadings are between 5 and 10 kg/yr with a median of 5.7 kg/yr) (Ecology and King County 2011). Total PBDE loading in Puget Sound from the three primary sources or pathways is 28 to 54 kg/yr or a median of 36.6 kg/yr (Ecology and King County 2011). Thus, PBDE loading from wastewater accounts for 25-38 percent of the total loading into Puget Sound (Ecology and King County 2011).

Figure 9 is a comprehensive map (produced by the People for Puget Sound 2008) of wastewater outfalls (municipal, industrial, and combined sewage overflow systems) in the Puget Sound. There are over 200 facilities or outfalls that discharge wastewater. Because outfalls from treatment plants are point sources for PBDEs, they can be a significant source to the local aquatic environment, including the water column and sediments. Marine water column sampling in Puget Sound showed PBDEs were routinely detected (Gries and Osterberg 2011). Concentration levels were almost 10 times higher in Puget Sound waters (51 to 18,700 pg/L, Gries and Osterberg 2011) than those reported in the southern Strait of Georgia (14.8 to 23.4, Dangerfield et al. 2007). Additionally, the range of detected total PBDEs in Puget Sound waters was much wider than the range of detected PCBs. Although PCBs have been detected in wastewater effluents, wastewater is currently not considered a significant source for PCBs (Grant and Ross 2002). In water column samples, PCBs were found in higher concentrations in deep water compared to shallow water, however, PBDEs had no apparent relationship with any other water column parameter examined (Gries and Osterberg 2011).

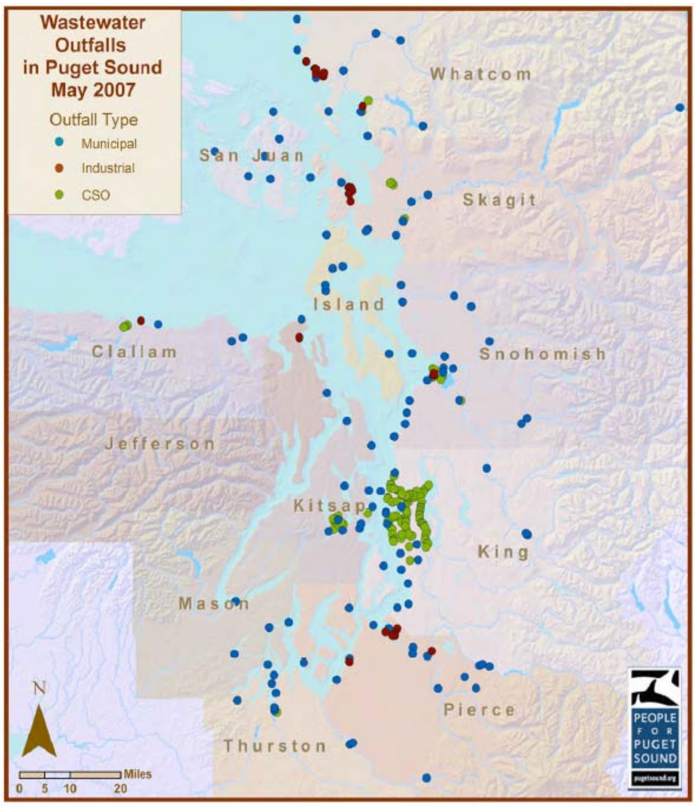

Figure 9. Outfall pipe locations from municipal sewage treatment plants, industrial facilities, and CSO, or combined sewage overflow systems (figure from People for Puget Sound 2008).

PBDEs in the Puget Sound Sediment. The Washington State Chemical Action Plan calls for baseline monitoring of PBDEs in Puget Sound because they are considered persistent bioaccumulative toxins. Marine sediment can act as a sink and sequester or bury contaminants, or the contaminated sediment can act as a source for aquatic food webs (i.e., a major pathway to killer whales). Since 2004, the Puget Sound Assessment and Monitoring Program (PSAMP) has measured PBDEs in the Puget Sound sediment (Dutch and Weakland 2009). Samples collected from 2004 to 2008 indicate that congeners -47, -99, and -209 were detected in the highest frequency and in highest concentrations in sediment near urban areas (Figure 10). Currently, PBDE concentrations in the sediment at the Solo Point outfall are unknown and the closest sampling point is Station 44, approximately 1 to 2 miles away. However, several studies have found higher PBDE concentrations in the sediments near wastewater outfalls (e.g., Gevao et al.

2006; Law et al. 2006; Samara et al. 2006; Johannessen et al. 2008; Grant et al. 2011). For example, measured PBDE concentrations in sediment immediately adjacent to the Iona Island wastewater outfall pipe (~12,700 pg/g) were 7 to almost 50 times greater than that measured elsewhere (Johannessen et al. 2008, see Figure 11). Therefore, it is likely that there are higher PBDE concentrations near the Solo Point wastewater treatment plant outfall than what is measured at Station 44 in Figure 10.

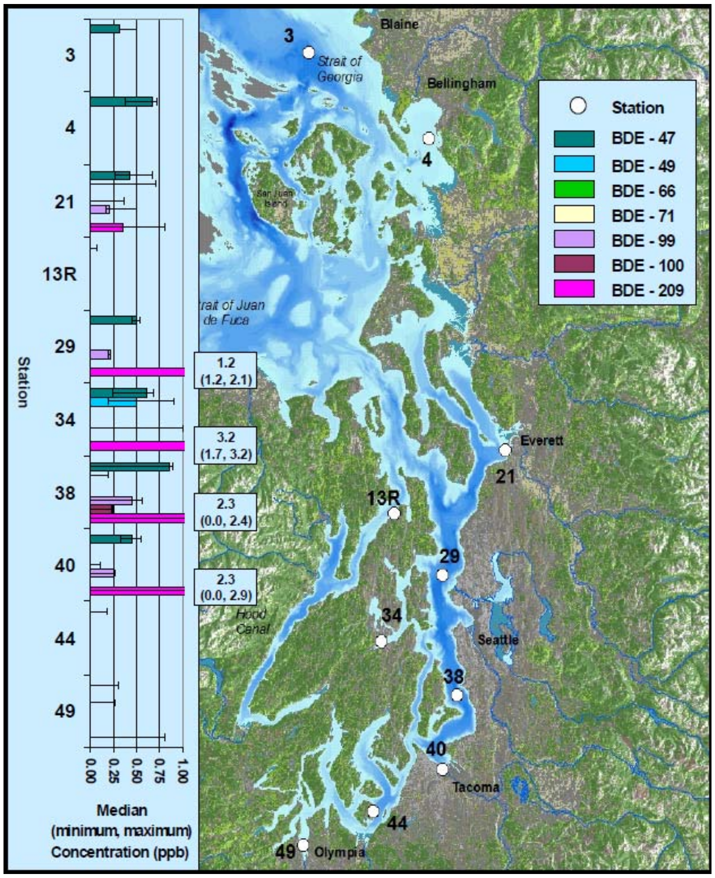

Figure 10. Concentrations of PBDE congeners detected at 10 PSAMP long-term sediment monitoring stations collected from Puget Sound in April, 2005. (Congeners BDE-138, -153, -154, -183, -184 were included in the analysis, but not detected in any sample). Reprinted from Dutch and Aasen (2007).

Sediment cores from the Strait of Georgia indicate PBDEs first entered the local aquatic environment around 1978 and have been rapidly increasing ever since (Johannessen et al. 2008). PBDEs in the Strait of Georgia sediment are strongly correlated to proximity to source and are increasing in concentration (Johannessen et al. 2008). This is unlike PCBs in surface sediment where sediment accumulation and mixing rates (more environmental processes) strongly influence concentration levels which are declining (Johannessen et al. 2008). Grant et al. (2011) examined PBDE and PCB patterns in surface sediment in the Strait of Georgia in the context of local sedimentation and contamination history. Total PBDE concentrations ranged between 87 and 12,700 pg/g. Hotspots were located near Victoria, Vancouver, and off the Campbell River and included high levels of both PBDEs and PCBs (see Figure 12). DFO (2010) predicted that there is an increase in the delivery of PCBs to killer whales when PCB concentration levels in in-water disposed dredged material are above ambient sediment levels. Although there are no predictions on PBDE delivery to killer whales, it is reasonable to assume that high levels of PBDEs in the sediment (or levels higher than ambient) can increase the delivery of PBDEs to killer whales, similar to predictions for PCBs.

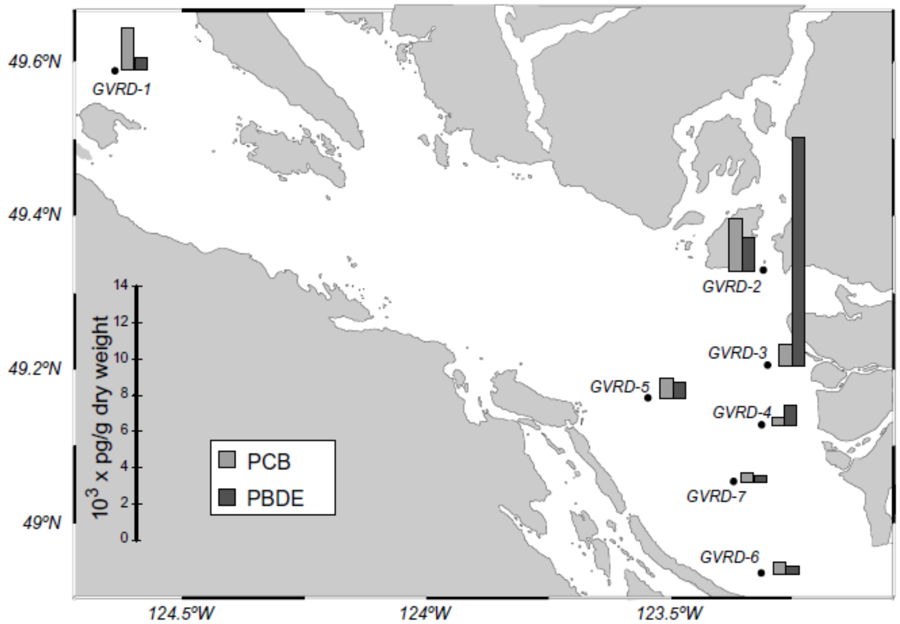

Figure 11. Distribution of total PCBs and total PBDEs in Strait of Georgia sediments (surface sediment concentrations). Site GVRD-3 is near the Iona Island wastewater outfall. Reprinted from Johannessen et al. (2008).

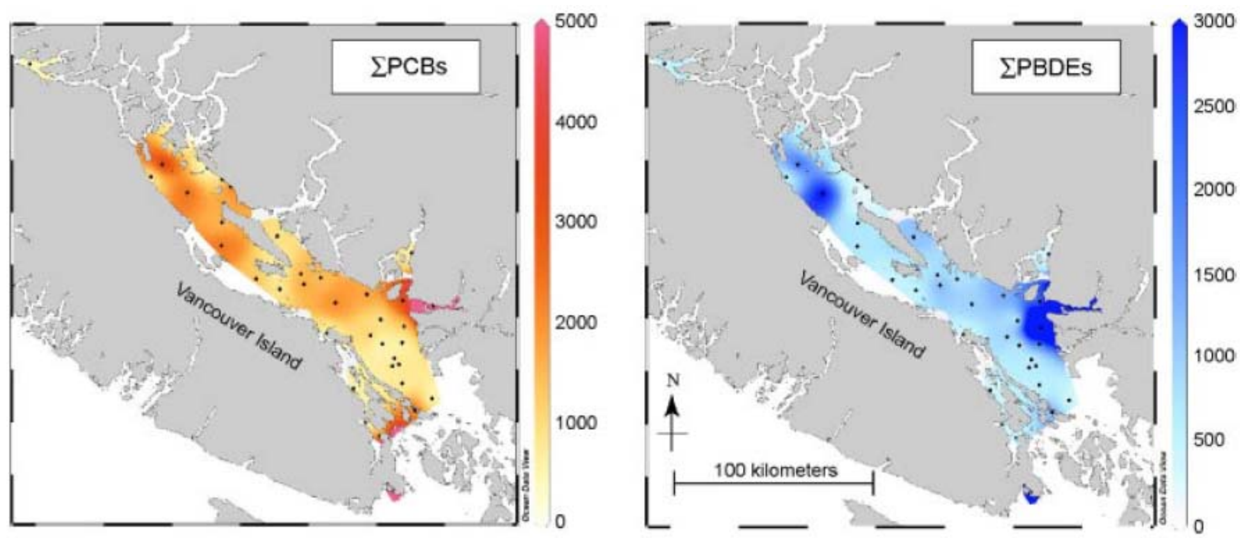

Figure 12. Contour plot of the concentrations of total PCBs and total PBDEs in surficial sediment samples collected from the Strait of Georgia, Canada. Reprinted from Grant et al. (2011).

Comparative PBDEs in Resident Killer Whales. Both the Southern and Northern Resident killer whale populations consume salmonids with Chinook being their primary prey (Ford et al. 1998;

Hanson et al. 2010c). Northern Resident killer whales occupy less industrialized waters than Southern Residents. In general, Southern Residents have significantly higher contaminant levels than the Northern Residents. For example, Rayne et al. (2004) observed significantly higher PBDE concentrations in male Southern Residents than in male Northern Residents. Elevated PCB levels were also measured in Southern Residents compared to their northern counterparts (Ross et al. 2000). These higher body burdens in Southern Resident killer whales may result from ingesting highly contaminated and localized prey in industrialized areas, such as the inland waters of Puget Sound and Georgia Strait (Ross et al. 2000; Rayne et al. 2004; Puget Sound Action Team 2007).

Prey Quality. The majority of growth in salmon occurs while feeding in saltwater (Quinn 2005). Therefore, the majority (greater than 96 percent) of POPs in adult salmon are accumulated while feeding in the marine environment (Cullon et al. 2009; O'Neill and West 2009). As discussed in the Status of the Species, the marine distribution is an important factor affecting POP accumulation as is evident in the different salmon populations (see Table 4). Although, Chinook salmon generally had higher concentrations of POPs than other Pacific salmon species, the levels varied considerably among Chinook populations, with those populations feeding in close proximity to land-based sources of contaminants having higher concentrations (O'Neill et al. 2006).

Adult salmon that feed in the Puget Sound are at a greater risk of exposure to contaminants than those that feed in the Strait of Georgia or the outer coast because of proximity to urban areas, the increased residence time of water (i.e., hydrological isolation) that can prolong exposure of POPs, and the highly contaminated pelagic food web (O'Neill and West 2009). For example, contaminant concentrations in Pacific herring from Puget Sound, a common prey item of Chinook salmon, are 3 to 9 times higher than herring from the Strait of Georgia (West et al. 2008). PCB concentrations in Chinook salmon from Puget Sound ranged from 10 to 220 ng/g, which was 3 to 5 times higher than the average PCB levels from 6 other populations along the outer coasts of Alaska, British Columbia, Washington, and Oregon (O'Neill and West 2009). O'Neill and West (2009) suggest that the wide range and higher levels of PCB concentrations (which were not observed in Chinook populations outside Puget Sound) are caused by the residency of some Chinook salmon in Puget Sound. Resident Chinook spend more time within the Puget Sound waters than conspecifics, which migrate out of Puget Sound and spend more time feeding off the outer coast. Thus, resident Chinook are likely exposed to more contaminants. O'Neill and West (2009) estimated that a considerable proportion of subyearlings and yearling out-migrants from Puget Sound displayed resident behavior, at least 29 percent and 45 percent, respectively. These estimates were considered conservative because they did not include any resident fish that may be caught during the July through September fisheries. In a separate study, Chamberlin (2009) found that 30 percent of all Puget Sound Chinook salmon over all years sampled displayed resident behavior.

O'Neill et al. (2004) measured PBDE concentrations in English sole, rockfish, lingcod, Pacific herring, and resident and migratory Chinook salmon to determine the extent that PBDEs have accumulated into the benthic and pelagic food webs in Puget Sound. Their preliminary results show that PBDEs are in both the pelagic and benthic food webs. O'Neill et al. (2004) also found the ratio between PBDEs and PCBs is higher in pelagic species than in benthic. Therefore, PBDEs are likely accumulating at a greater degree in pelagic species than they are in benthic species. Similar to PCBs, PBDEs were higher in resident Chinook salmon than in Chinook salmon that migrate out of the Puget Sound, suggesting that Puget Sound is a predominant source for PBDEs.

Incremental Increase Model. An incremental PBDE accumulation model was developed to assess the degree to which PBDE loadings affect Southern Resident PBDE body burdens. We assumed for our model simulations that all three pods in the Southern Resident population consumed solely Chinook salmon (Chinook salmon reflect the whales' average diet). Although this is a simplified diet, research on Southern Resident killer whales indicates that Chinook are the predominant prey species (Ford et al. 1998; Ford & Ellis 2006; Ford et al. 2010; Hanson et al. 2010c) and estimated contaminant levels in the whales using this assumption have been accurately predicted (Mongillo 2009). Two scenarios were evaluated, the first scenario included PBDE input from all sources except the estimated PBDE loadings from the Solo Point wastewater effluent[4]. The second scenario included the estimated PBDE loadings from all sources including PBDEs from Solo Point wastewater effluent, and is the focus of the Effects section. These two scenarios were performed in order to isolate the input of PBDEs from the Solo Point WWTP that are expected as a result of the proposed action, and to put the contribution from Solo Point in the context of other wastewater inputs. Because contaminants are a range-wide threat to the status of the species, the effects of the proposed action are not easily separable from baseline and cumulative effects. Therefore, we characterized PBDE accumulation across the year, which includes accumulation both inside and outside the action area. While such accumulation is an effect of activities that are part of the environmental baseline as well as cumulative effects, this analysis is placed in this section to maximize logical flow. Below is a brief description of the steps involved in the incremental PBDE accumulation model, followed by subsections that consist of more detail on each step.

The primary steps in the incremental PBDE accumulation model are: Estimate the proportion of the whales' diet composed of migratory and resident Chinook; Estimate the whales' food energy needs and food ingestion rates in both coastal and inland waters; Estimate the total PBDEs ingested via prey consumption of migratory and resident Chinook salmon; Project PBDE concentrations in individual whales with and without the fraction of PBDEs from the Solo Point wastewater effluent.

Step 1: Proportion of diet composed of migratory and resident Chinook. The first step in the model was to estimate the amount of time the Southern Residents spend in the coastal and inland waters to estimate the proportion of the whales' diet that consists of migratory and resident Chinook salmon. It is assumed that when the whales are in coastal waters (west of the Strait of Juan de Fuca), they are consuming the migratory Chinook salmon and not the resident Chinook

[4] The vast majority of wastewater treatment plants are non-federal. While the future inputs of these plants are cumulative effects, not part of the baseline, their effects in the context of the incremental accumulation model are considered here to set the stage for the description of the effects of the Solo Point facility in the Effects section. The inputs from these non-federal facilities are considered in combination with those from federal facilities because it is not possible to tease out PBDE loadings resulting from the operation of federal wastewater facilities from those resulting from non-federal facilities. However, because the vast majority of inputs are from non-federal facilities, the inclusion of inputs from federal facilities, which are part of the baseline, are not likely to noticeably alter the results.

salmon, whereas when they are in inland waters (i.e., their designated critical habitat) they are likely consuming Chinook salmon roughly proportional to the local Chinook stock abundance (this assumption is based on the findings in Hanson et al 2010c).

Hanson and Emmons (2010) provided a compilation of Southern Resident killer whale sightings specific to each pod in inland waters (January 2003 to December 2009, Table 2 in the Status of the Species). For purposes of this analysis, we assumed that Southern Residents occurred west of the Strait of Juan de Fuca (in coastal waters) on days they were not sighted in inland waters, primarily because the population is highly visible in inland waters. The average number of days in a year J, K, and L pods spend in inland waters is 166 days, 97 days, and 87 days, respectively. Therefore, the average number of days in a year they spend in coastal waters is 199 days, 268 days, and 278 days, respectively.

As described above, at least 30 percent of Puget Sound origin Chinook salmon remain in Puget Sound and become resident fish. Harrison fish, the most abundant population from the Fraser River, have a more coastal distribution (DFO 1999), but are also believed to be somewhat resident in the Salish Sea, however, there are no estimates of the percent residency. Therefore we assume that when in inland waters, Southern Resident killer whales consume 70 percent migratory and 30 percent resident Chinook salmon (i.e., in general proportion to the stocks availability). Table 7 provides the assumed diet for each pod based on their distribution.

Table 7. Proportion of time spent in a year for each pod is estimated from Hanson and Emmons, 2010, unpubl. report (see Table 2 in Status of the Species). The diet is assumed 100% Chinook salmon and is consumed in roughly the proportion that the stocks are available.

	Proportion of Time Spent		Proportion of Diet	
Pods	Coastal Waters	Inland Waters	Migratory	Resident
J	55%	45%	55% + (45% * 70%) = 86.5%	45% * 30% = 13.5%
K	73%	27%	73% + (27% * 70%) = 91.9%	27% * 30% = 8.1%
L	76%	24%	76% + (24% * 70%) = 92.8%	24% * 30% = 7.2%

Step 2: The Whales' Food Energy Needs. We assessed the whales' food energy needs from Chinook using the best available information on their metabolic needs, time spent in inland waters, and the caloric content of the prey. Noren (2011) developed estimates of the potential range of daily energy expenditure and prey energy requirements for Southern Resident killer whales for all ages and both sexes. NMFS used this information to estimate the maximum energetic requirements per year in coastal and inland waters for each age- and sex-class for each pod of the Southern Resident population (Table 8).

We focused on the maximum estimates for several reasons. The maximum and minimum field metabolic rates (FMRs, or daily energy expenditure) reported by Noren (2011) fall within the range of FMRs of wild killer whales, based on daily activity budgets. Thus, the maximum of this reported range from Noren (2011), used in this biological opinion, represents realistic values for wild killer whales. The FMRs and resulting calculated daily prey energy requirements from Noren (2011) do not account for the increased energetic cost of body growth in juvenile whales or the increased cost of lactation in females who are nursing calves. Although the costs of these physiological processes are not exactly known, they could increase the daily prey energy requirements (DPER) of specific individuals that fall within these categories. For example, prey consumption rates in lactating females can increase 1.5 to 2 times over consumption rates of non-lactating females (Kriete 1995; Kastelein 2002; Kastelein et al. 2003a, b). By using the maximum daily prey energy requirements, our calculations are more likely to account for energetic costs in the population that were not included by Noren (2011), than if we used the minimum daily prey energy requirements.

We computed the energy requirements for each individual based on its age, sex, and pod membership, and multiplied the daily energy requirements of each individual by the number of days in a year that the pod was assumed to be in coastal and inland waters (Table 7). We provide all DPERs (for each age and sex class) in Table 8 for the projected values. The model results, however, focus on two individuals with known PBDE concentrations in the blubber. The focus on these two individuals was based on their higher likelihood of exposure (they are from J pod, which, in general, spend more time in the inland waters than K and L pods), and because they are at the highest risk for contaminant-induced toxicity (e.g., adult males have relatively higher body burdens and calves are exposed to high concentrations of contaminants during a critical period of development).

Table 8. Chinook energy requirements or daily prey energy requirements (DPER) for Southern Resident killer whales in each age and sex class in coastal and inland waters in kilocalories per year (Kcal/yr).

Age- and Sex-Class	Coastal DPER (Kcal/yr)			Inland DPER (Kcal/yr)		
	J pod	K pod	L pod	J pod	K pod	L pod
Immature/juvenile age 1	9,881,743	13,322,264	13,818,834	8,243,062	4,802,541	4,305,971
Immature/juvenile age 2	13,357,875	18,008,679	18,679,929	11,142,750	6,491,946	5,820,696
Immature/juvenile age 3	16,873,210	22,747,946	23,595,846	14,075,140	8,200,404	7,352,504
Immature/juvenile age 4	20,220,788	27,261,048	28,277,168	16,867,592	9,827,332	8,811,212
Immature/juvenile age 5	23,248,374	31,342,747	32,511,007	19,393,116	11,298,743	10,130,483
Immature/juvenile age 6	25,918,954	34,943,141	36,245,601	21,620,836	12,596,649	11,294,189
Immature/juvenile age 7	28,186,360	37,999,989	39,416,389	23,512,240	13,698,611	12,282,211
Immature/juvenile age 8	30,076,064	40,547,630	42,058,990	25,088,576	14,617,010	13,105,650
Immature/juvenile age 9	31,635,428	42,649,917	44,239,637	26,389,352	15,374,863	13,785,143
Immature/juvenile age 10	32,892,511	44,344,677	45,997,567	27,437,974	15,985,808	14,332,918
Immature/juvenile age 11	33,901,441	45,704,886	47,408,476	28,279,594	16,476,149	14,772,559
Immature/juvenile age 12	34,701,620	46,783,663	48,527,463	28,947,080	16,865,037	15,121,237
Young adult female age 13	35,817,612	48,288,209	50,088,089	29,878,008	17,407,411	15,607,531
Adolescent male age 13	37,229,915	50,192,233	52,063,083	31,056,110	18,093,792	16,222,942
Young adult female age 14	36,922,062	49,777,195	51,632,575	30,799,308	17,944,175	16,088,795
Adolescent male age 14	39,702,291	53,525,415	55,520,505	33,118,494	19,295,370	17,300,280
Young adult female age 15	38,015,766	51,251,693	53,162,033	31,711,644	18,475,717	16,565,377
Adolescent male age 15	42,124,121	56,790,452	58,907,242	35,138,714	20,472,383	18,355,593
Young adult female age 16	39,099,122	52,712,241	54,677,021	32,615,348	19,002,229	17,037,449
Adolescent male age 16	44,500,579	59,994,320	62,230,530	37,121,086	21,627,345	19,391,135
Young adult female age 17	40,172,528	54,159,374	56,178,094	33,510,752	19,523,906	17,505,186
Adolescent male age 17	46,835,446	63,142,116	65,495,656	39,068,764	22,762,094	20,408,554
Young adult female age 18	41,236,382	55,593,629	57,665,809	34,398,188	20,040,941	17,968,761
Adolescent male age 18	49,132,105	66,238,401	68,707,351	40,984,570	23,878,274	21,409,324
Young adult female age 19	42,291,281	57,015,812	59,141,002	35,278,154	20,553,623	18,428,433
Adolescent male age 19	51,393,541	69,287,200	71,869,790	42,870,994	24,977,335	22,394,745
Adult female age ≥20	43,337,225	58,425,921	60,603,671	36,150,650	21,061,954	18,884,204
Adult male age ≥20	53,622,142	72,291,732	74,986,312	44,730,028	26,060,438	23,365,858

To estimate the available energy from Chinook, we applied a regression to convert mass (in kilograms, kg) to kilocalories (kcals) (Figure 13 O'Neill et al. in prep). The regression is based on data available on proximate composition of individual Chinook from different locations, and with different lipids and proteins (for detailed methods see O'Neill et al. in prep.). Puget Sound Chinook contain, on average, lower lipid content than Skeena Chinook. Therefore, a Puget Sound Chinook of a specific size would have lower kcal content than a comparable size fish from the Skeena. In general, populations will differ in their lipid content depending upon the length and elevation of their upriver migration. Within a population, lipid content will vary with maturation condition among individuals. Additionally, each data point on the regression represents a composite of 3 fish. Therefore, the kg to kcal relationship for each population may be more variable than is shown in the regression (i.e., more representative of an average value). Estimates of Chinook kilocalories based on this relationship are likely to be lower than the energy of prey available to Southern Residents, because this effort sampled Chinook in terminal areas whereas the whales have access to the Chinook in the ocean (i.e., before the Chinook have expended energy traveling to terminal destinations).

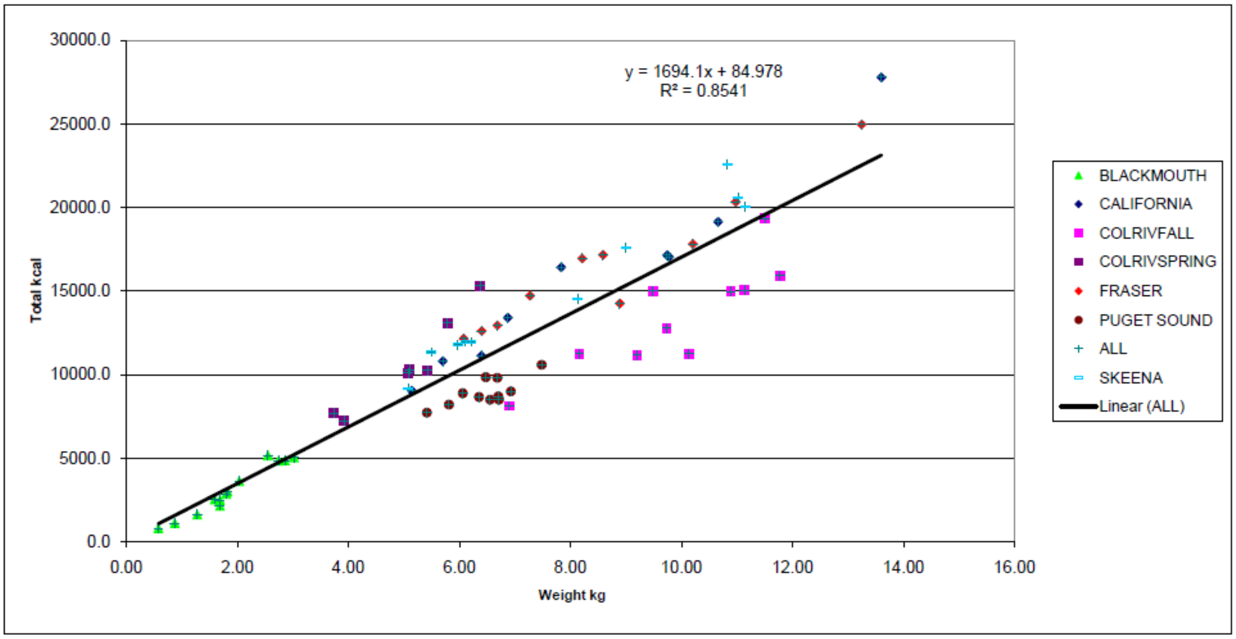

Figure 13. Chinook salmon total kilocalories (kcal) per total weight in kilograms (kg) for several Chinook salmon stocks including: resident Chinook (BLACKMOUTH), California, Columbia River fall-run (COLRIVFALL), Columbia River spring-run (COLRIVSPRING), Fraser River, Puget Sound, and Skeena River.

To estimate the food ingestion rate (FIR), we multiplied the kilogram to kilocalorie ratio (1 kg / 1779 kcal) derived from this regression to the maximum annual prey energetic requirement for each individual whale based on its age, sex, pod membership, and time in coastal and inland waters (Table 9). This provided an estimate of FIR in kg of Chinook per year for each age and sex class in both coastal and inland waters.

Table 9. Coastal and inland food ingestion rates (FIR) in kg/yr for each age and sex class and for each pod of the Southern Resident killer whales.

Age- and Sex-Class	Coastal FIR (kg/yr)			Inland FIR (kg/yr)		
	J pod	K pod	L pod	J pod	K pod	L pod
Immature/juvenile age 1	5,555	7,489	7,768	4,634	2,700	2,420
Immature/juvenile age 2	7,509	10,123	10,500	6,263	3,649	3,272
Immature/juvenile age 3	9,485	12,787	13,264	7,912	4,610	4,133
Immature/juvenile age 4	11,366	15,324	15,895	9,482	5,524	4,953
Immature/juvenile age 5	13,068	17,618	18,275	10,901	6,351	5,694
Immature/juvenile age 6	14,569	19,642	20,374	12,153	7,081	6,349
Immature/juvenile age 7	15,844	21,360	22,156	13,217	7,700	6,904
Immature/juvenile age 8	16,906	22,792	23,642	14,103	8,216	7,367
Immature/juvenile age 9	17,783	23,974	24,868	14,834	8,642	7,749
Immature/juvenile age 10	18,489	24,927	25,856	15,423	8,986	8,057
Immature/juvenile age 11	19,056	25,691	26,649	15,896	9,261	8,304
Immature/juvenile age 12	19,506	26,298	27,278	16,272	9,480	8,500
Young adult female age 13	20,134	27,143	28,155	16,795	9,785	8,773
Adolescent male age 13	20,927	28,214	29,265	17,457	10,171	9,119
Young adult female age 14	20,754	27,980	29,023	17,313	10,087	9,044
Adolescent male age 14	22,317	30,087	31,209	18,616	10,846	9,725
Young adult female age 15	21,369	28,809	29,883	17,826	10,385	9,312
Adolescent male age 15	23,679	31,923	33,113	19,752	11,508	10,318
Young adult female age 16	21,978	29,630	30,735	18,334	10,681	9,577
Adolescent male age 16	25,014	33,724	34,981	20,866	12,157	10,900
Young adult female age 17	22,582	30,444	31,578	18,837	10,975	9,840
Adolescent male age 17	26,327	35,493	36,816	21,961	12,795	11,472
Young adult female age 18	23,180	31,250	32,415	19,336	11,265	10,100
Adolescent male age 18	27,618	37,234	38,621	23,038	13,422	12,034
Young adult female age 19	23,773	32,049	33,244	19,830	11,553	10,359
Adolescent male age 19	28,889	38,947	40,399	24,098	14,040	12,588
Adult female age ≥20	24,360	32,842	34,066	20,321	11,839	10,615
Adult male age ≥20	30,142	40,636	42,151	25,143	14,649	13,134

Step 3: PBDEs Ingested via Prey Consumption from Coastal and Inland Waters. The third step was to estimate the amount of PBDEs ingested by the whales based on the estimated proportion of migratory and resident Chinook consumed using the measured PBDE concentrations in Chinook salmon. On average, PBDE concentrations in resident Chinook salmon were 40 ng/g wet weight (O'Neill et al. 2006). For the first scenario, where there is no PBDE contribution from the Solo Point outfall, this value would be slightly lower. Based on the data summarized in Table 4, the average PBDE concentration in Chinook salmon from Alaska to California (i.e., primarily migratory salmon) was 6.22 ng/g wet weight.

The migration distribution patterns of resident Chinook salmon in the Puget Sound have not been adequately evaluated. Therefore, we assumed resident Chinook are equally distributed throughout Puget Sound and have equal likelihood of exposure to the Solo Point effluent. We also assumed contaminant levels in resident Chinook change in the same proportion as the fraction of change in total PBDE loadings in Puget Sound. For example, if the total PBDE loadings in Puget Sound were reduced by 10 percent, we would assume the resident Chinook experienced a 10 percent reduction of PBDE concentration as well.

The PBDE intake was estimated by the product of the PBDE concentration in the prey and the amount of prey consumed, of which a proportion is assimilated and eliminated or excreted. Currently, there are no assimilation rates or elimination rates for PBDEs in whales. There is evidence that bottlenose dolphins (*Tursiops truncates*) have an almost 100 percent absorption efficiency of organochlorines (Marsili et al. 1995). Furthermore, Hickie et al. (2007) estimated a relatively small elimination rate for PCBs in killer whales, at approximately 1.5 percent. Elimination rates likely vary among species, individuals, and chemicals. We assumed 100 percent assimilation and zero elimination because of the lack of data specific to PBDEs in killer whales and because of the potential high degree of assimilation and relatively low elimination rate of similar persistent pollutants. The PBDE concentrations in the migratory and resident Chinook were multiplied by the coastal and inland food ingestion rates (FIRs) for each age- and sex-class in each pod to estimate the total PBDE load consumed by an individual whale in each year of its life.

Step 4: PBDEs in the Whales. We estimated PBDE intake and accumulation in two individual whales that have known existing PBDE concentrations (Krahn et al 2007, 2009), have the highest probability of being exposed to contaminants from the effluent (i.e., individuals from J pod because they spend more time in inland waters than K or L pod), and are believed to be at highest risk for contaminant-induced toxicity (e.g., calves and adult males because they have higher body burdens and calves are exposed during a sensitive period of development and growth). Biopsy samples were obtained from two male individuals (age 3 and 15) from J pod and blubber PBDE concentrations were measured (Krahn et al. 2007, 2009). The PBDE levels in the calf and the adult male were measured at 15,000 ng/g lipid, and 6,300 ng/g lipid, respectively. Thus, the first year of the model simulation reflected these measured existing concentrations in the whales. The second year of the model projection was for a 4 year-old juvenile and a 16 year-old adult male. The FIR for a 4 year-old juvenile in coastal and inland waters was estimated to be 11,366 kg/yr and 9,482 kg/yr, respectively (see Table 9). These ingestion rates were multiplied by the PBDE concentrations in the prey to estimate the PBDE intake for that year. For example, the coastal FIR was multiplied by the PBDE burden in migratory Chinook salmon. The inland FIR was multiplied by the PBDE burden in migratory Chinook salmon (where 70 percent of fish were migratory), and 30 percent of the inland FIR was multiplied by the PBDE burden in the resident Chinook salmon. These values were summed together to provide a total PBDE intake in the whales for each projected year.

Many congeners (or forms) of PBDEs resist metabolic degradation and accumulate in individuals throughout their lives. We evaluated a 20 year time period in order to have a biologically meaningful analysis for a long-lived species that accumulates persistent pollutants. This time frame is meant to represent recent PBDE body burdens in the whales. We do not evaluate beyond this time frame because assuming that PBDE inputs into Puget Sound (and thus PBDE levels in Chinook salmon) beyond 20 years from present would remain similar to those found currently would be too speculative. As discussed above, state laws regarding PBDE use are evolving, thus PBDE concentrations in Chinook salmon could change significantly in the future. Figure 14 displays the PBDE body burdens in two J pod individuals in the first scenario. The model predicted that the calf experiences a growth dilution where he was growing faster than his intake of contaminants and thus his concentration declines during those years of growth. It was only after he completed growing that the accumulation increased. The model predicted that the adult male experiences a continued increase in accumulation of PBDEs.

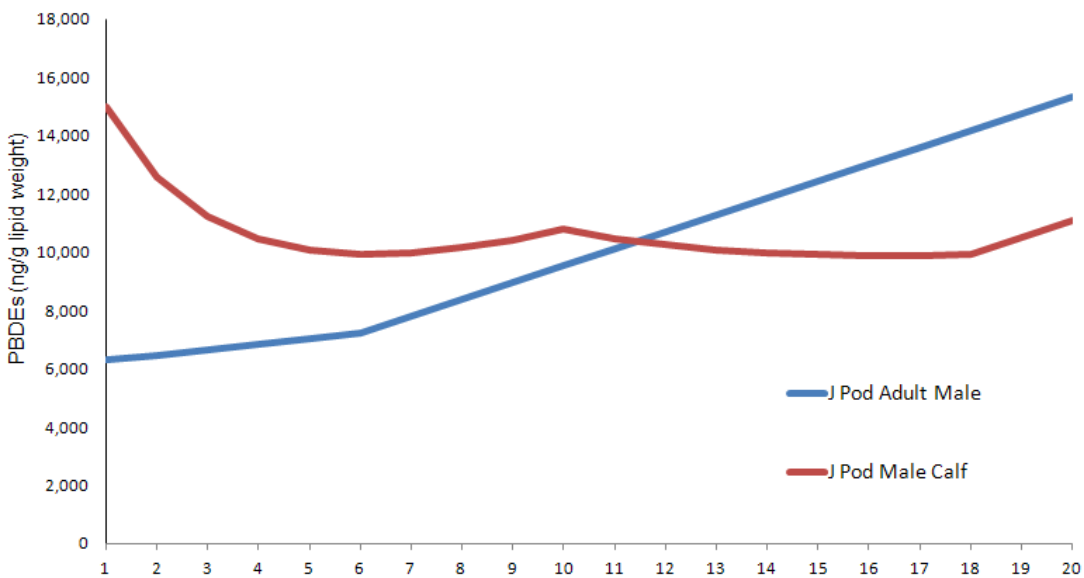

Figure 14. Projected PBDE concentrations (in ng/g lipid blubber weight) for two individuals that have measured PBDE concentration values for the initial year (Krahn et al. 2007, 2009). The adult J pod male was 15 when sampled and the J pod calf was sampled at 3 years of age.

Prey Availability. When prey is scarce, whales likely spend more time foraging than when it is plentiful. Increased energy expenditure and prey limitation can cause nutritional stress. Nutritional stress is the condition of being unable to acquire adequate energy and nutrients from prey resources and as a chronic condition can lead to reduced body size and condition of individuals and lower birth and survival rates in a population. Ford et al. reported correlated declines in both the Southern Resident killer whales and Chinook salmon and suggested the potential for nutritional stress in the whales (Ford et al. 2005; Ford et al. 2010). Food scarcity could also cause whales to draw on fat stores, mobilizing contaminants stored in their fat and potentially have the ability to alter thyroid homeostasis, reduce immune function, cause neurotoxicity, reproductive failure, and restrict the development and growth of the individual (as discussed above). Thus, nutritional stress may act synergistically with high contaminant burdens in the whales and result in contaminant-induced adverse health effects, higher mortality rates, or lower birth rates.

The availability of Chinook to Southern Resident killer whales is affected by a number of natural and human actions. Climate effects from Pacific decadal oscillation and the El Nino/Southern oscillation conditions and events cause changes in ocean productivity that can affect natural mortality of salmon. Predation in the ocean also contributes to natural mortality of salmon. Salmonids are prey for pelagic fishes, birds, and marine mammals (including Southern Residents). The most notable human activities that cause adverse effects include land use activities that result in habitat loss and degradation, hatchery practices, overharvest and hydropower systems. Details regarding baseline conditions of Puget Sound Chinook salmon in inland waters that are listed under the Endangered Species Act are in the subsections above. The baseline also includes activities affecting Chinook that are not ESA-listed (e.g., some of the Puget Sound hatchery Chinook stocks are not part of the listed entity), as well as Fraser River and Georgia Strait stocks of Chinook.

Vessel Activities and Sound. Vessels used for a variety of purposes (commercial shipping, military, recreation, fishing, whale watching and public transportation) occur in inland waters of the Southern Residents' range. Several studies in inland waters of Washington State and British Columbia have linked interactions of vessels and Northern and Southern Resident killer whales with short-term behavioral changes (Kruse 1991; Williams et al. 2002a, 2002b; Foote et al. 2004; Bain et al. 2006; Noren et al. 2009a; Noren et al. 2009b; Holt 2008; Holt et al. 2009; Noren et al. 2010; Noren et al. in press). These vessel activities may affect foraging efficiency, communication, and/or energy expenditure through the physical presence of the vessels, underwater sound created by the vessels, or both. Collisions of killer whales with vessels are rare, but remain a potential source of serious injury and mortality.

Vessel sounds in inland waters are from large ships, tankers and tugs, as well as from whale watch vessels, ferries and smaller recreational vessels. Sound generated by large vessels is a source of low frequency (5 to 500 Hz) human-generated sound in the world's oceans (NRC 2003). While larger ships generate some broadband noise in the hearing range of whales, the majority of energy is below their peak hearing sensitivity. Such vessels do not target whales, move at relatively slow speed and are likely detected and avoided by Southern Residents. Commercial sonar systems designed for fish finding, depth sounding, and sub-bottom profiling are widely used on recreational and commercial vessels and are often characterized by high operating frequencies, low power, narrow beam patterns, and short pulse length (NRC 2003). Frequencies fall between 1 and 500 kHz, which is within the hearing range of some marine mammals including killer whales and may have masking effects (i.e., sound that precludes the ability to detect and transmit biological signals used for communication and foraging).

In inland waters, the majority of vessels in close proximity to the whales are commercial and recreational whale watching vessels and the average number of boats accompanying whales can be great during the summer months (i.e., from 1998 to 2010 an average of about 15 to 20 boats were within ½ mile of the whales in inland waters from May to September; Koski 2010). Sound generated from whale watch vessels varies by vessel size, engine type, and operating speed (Holt 2008). A few studies have evaluated the consequences of short-term behavioral responses on the health of the cetacean populations (i.e., Williams et al. 2006; Noren et al. 2009b; Holt et al. 2009; Lusseau et al. 2009). Likely effects of vessel interaction and noise include increased energy expenditure from behavioral responses and decreased foraging efficiency due to masking. Both of these effects, particularly in combination, may reduce killer whale fitness. NMFS recently issued vessel management regulations to protect Southern Resident killer whales from vessel effects. These regulations were effective May 16, 2011 (76 FR 20870; April 14, 2011).

Non-Vessel Sound. Anthropogenic (human-generated) sound in inland waters is generated by other sources besides vessels, including construction activities and military operations. Natural sounds in the marine environment include wind, waves, surf noise, precipitation, thunder, and biological noise from other marine species. The intensity and persistence of certain sounds (both natural and anthropogenic) in the vicinity of marine mammals vary by time and location and have the potential to interfere with important biological functions (e.g., hearing, echolocation, communication).

In-water construction activities are permitted by the Army Corps of Engineers (ACOE) under section 404 of the Clean Water Act and section 10 of the Rivers and Harbors Act of 1899 and by the State of Washington under its Hydraulic Project Approval (HPA) program. NMFS conducts consultations on these permits and helps project applicants incorporate conservation measures to minimize or eliminate potential effects of in-water activities, such as pile driving, to marine mammals. Sound, such as sonar generated by military vessels also has the potential to disturb killer whales.

Oil Spills. Oil spills have occurred in the range of Southern Residents in the past, and there is potential for spills in the future. Oil can be discharged into the marine environment in any number of ways, including shipping accidents, at refineries and associated production facilities, and pipelines. Despite many improvements in spill prevention since the late 1980s, much of the region inhabited by Southern Residents remains at risk from serious spills because of the heavy volume of shipping traffic and proximity to petroleum refining centers in inland waters. Numerous oil tankers transit through the inland waters range of Southern Resident killer whales throughout the year. The magnitude of risk posed by oil discharges in the action area is difficult to precisely quantify, but the volume of spills is decreasing (i.e., seven year comparison 2001-2007, for Seattle-Sector USCG, Smith unpubl. data). New oil spill prevention procedures in the state of Washington likely positively contribute to the decrease in spill volume (WDOE 2007).

Repeated ingestion of petroleum hydrocarbons by killer whales likely causes adverse effects; however, long-term consequences are poorly understood. In marine mammals, acute exposure to petroleum products can cause changes in behavior and reduced activity, inflammation of the mucous membranes, lung congestion, pneumonia, liver disorders, neurological damage (Geraci and St. Aubin 1990), potentially death and long-term effects on population viability (Matkin et al. 2008). In addition, oil spills have the potential to adversely impact habitat and prey populations, and, therefore, may adversely affect Southern Resident killer whales by reducing food availability.

NPDES Permits that have Undergone Endangered Species Act Section 7 Consultation. Discharged effluent from wastewater treatment plants can enter the food web, reduce the quality of prey available to Southern Resident killer whales, and increase the toxic chemicals in the whales. EPA delegates the majority of NPDES permits to the Department of Ecology. As discussed above, there are nearly 1000 municipal and industrial wastewater discharges into the Puget Sound that are permitted by the Department of Ecology. However, one recent NPDES permit action at a federal facility had a federal nexus and underwent section 7 consultation. That action met the standard of not jeopardizing the continued existence of the listed salmonids and killer whales or adversely modifying their critical habitat (NMFS 2010). NMFS provided the U.S. EPA with an informal consultation for the reissuance of the NPDES permit for Naval Air Station Whidbey Island Wastewater Treatment Plant (WWTP). In contrast to the Solo Point WWTP that discharges continuously, the Naval Air Station Whidbey Island WWTP releases effluent intermittently every day (approximately 3.75 hours/day), a discharge of 0.37 mgd. Due to the relatively small size, poor habitat quality and intermittent discharge of the mixing zone, NMFS anticipated that juvenile and adult salmonids would spend little if any time within area. Thus, any potential for fish exposed to the mixing zone to uptake contaminants and metals and any subsequent chance of bioaccumulation in the Southern Residents was determined to be extremely unlikely and therefore discountable.

Summary of Environmental Baseline for Southern Resident Killer Whales. Southern Resident killer whales are exposed to a wide variety of impacts in the action area from past and present state, federal or private actions and other human activities, as well as federal projects that have already undergone formal section 7 consultation, and state or private actions that are contemporaneous with this consultation. All of the activities discussed in the above section are likely to have some level of impact on Southern Resident killer whales when they are in the action area.

No single threat has been identified as the cause of the recent decline of the Southern Resident killer whales, although the three primary threats are identified as prey availability, environmental contaminants, and vessel effects and sound. Although it is not clear which threat or threats are most significant to the survival and recovery of Southern Residents, all of the threats identified are important to address. It is likely that multiple threats are acting together. For example, food availability is strongly associated with the variation in lipid content in an individual (Aguilar 1987). When a whale experiences nutritional stress, contaminants bound to lipids in the blubber can become mobilized and enter into circulation in the body. Once in circulation, these contaminants have the ability to alter thyroid homeostasis, reduce immune function, cause neurotoxicity, reproductive failure, and restrict the development and growth of the individual. Thus, nutritional stress may act synergistically with high contaminant burdens in the whales and result in contaminant-induced adverse health effects. The small size of the population increases the level of concern about all of these risks (NMFS 2008a).

2.4 Effects of the Action on the Species and its Designated Critical Habitat

"Effects of the action" means the direct and indirect effects of an action on the species or critical habitat, together with the effects of other activities that are interrelated or interdependent with that action, that will be added to the environmental baseline (50 CFR 402.02). Indirect effects are those that are caused by the proposed action and are later in time, but still are reasonably certain to occur. No interrelated or interdependent activities were identified by the EPA in the course of this consultation.

2.4.1 Effects on Listed Species

Based on the proposed permit limits discussed in the project description, review of monitoring data collected over the past permit cycle, and on reports from other WWTP's that similarly process human waste, NMFS expects constituents to be discharged at Solo Point to include metals, chlorine, nutrients, and toxicants for which no water quality criteria are established such as polycyclic aromatic hydrocarbons, phlalates, PBDE's, and personal care products that may have estrogenic or other physiological or toxicological effects. The following sections detail species-specific responses anticipated from exposure to constituent elements of the effluent the action area over the five year life of the NPDES permit. It will also examine the indirect effects on the listed species that occur when the prey species of the listed species are exposed to the constituent elements of the effluent over the same time period.

Effects for all Listed Puget Sound Salmonids

The proposed action will permit, for another five years, the continued discharge of treated wastewater effluent into the nearshore environment of South Puget Sound. Juvenile Puget Sound Chinook salmon and Puget Sound steelhead utilize the action area and will experience exposure to pollutants (from the WWTP effluent) in the water column and indirect exposure by consumption of food organisms, which have accumulated the pollutants (McCain et al. 1990). Nearly all juvenile Chinook salmon make some use, to varying degrees, of the Puget Sound nearshore. When small, they frequent the shallow water along the shoreline. As they grow and increase in size, the depth of water and diversity of habitats they use expand. Generally, as size increases they tend to move into deeper more offshore habitats.

Contaminants from Permitted Discharges. LaLiberte and Ewing (2006) observed that toxic contamination of water, sediments and organisms in the Puget Sound region, caused in substantial part by National Pollutant Discharge Elimination System (NPDES) permitted discharges, is widespread and likely to be harmful to Chinook salmon. They concluded that outfall mixing zone dilutions were routinely over-estimated for a significant number of Ecology NPDES permits, indicating that concentrations of contaminants remain higher, over a smaller area, rather than dissipating as anticipated over a broader zone and shorter timeframe. A systematic error that can occur in permit evaluations is the omission of accounting for tidal conditions that return previously discharged effluent back into the outfall mixing zone area (LaLiberte & Ewing 2006). This tidal return rate has the effect of raising background concentrations of effluent pollutants and significantly reduces dilution. Additionally, ambient velocities in the Puget Sound are typically overestimated in Ecology's NPDES permit evaluations also resulting in inflated rates of dilution. This leads Ecology to assume in many instances, that there are no undesirable effects from discharge of toxic chemicals and other contaminants, when in fact data have not been obtained to sufficiently support this conclusion.

Over-estimated dilution can result in less monitoring that could otherwise provide for adaptive management, fewer effluent limitations, and less wastewater treatment being required in the permits. Inaccurate and underestimation of effluent dilution factors create harm to Puget Sound organisms by distorting the results of the reasonable potential analysis as performed under EPA and NPDES permit requirements and guidance (LaLiberte & Ewing 2006). Also, mixing zones are calculated on a chemical-by-chemical basis rather than by looking at synergistic or cumulative impacts (Trim et al. 2008). The exact nature of the combined toxic effects may be additive or multiplicative (LaLiberte & Ewing 2006).

LaLiberte and Ewing (2006) also found that the Whole Effluent Toxicity (WET) testing approved by Ecology routinely allows minimal effluent concentrations to be used when evaluating an organism's exposure to total effluent effects that may not be representative of actual exposure conditions. Minimal effluent concentrations which are derived based on inflated dilutions and resulting from inaccurate mixing zone analyses as described above mean that where listed salmonids are present in contaminated areas, they can be exposed to higher concentrations over longer periods of time than accounted for, though these conditions exist in a more confined area. Moreover, WET testing is routinely conducted for a maximum of seven days, and therefore fails to account for longer-term effects, such as bioaccumulation, exhibited by many of the toxic chemicals.

Sub-lethal Effects of Exposure to Permitted Discharges. Effluent discharged from the Solo Point WWTP is unlikely to cause fish kills of documentable scale from acute exposure to the concentrations of pollutants released.Of greater concern are the chronic low level concentrations of contaminants which do not cause acute fish kills but still may impact populations through increased disease (immunosuppression) or through decreased predator avoidance abilities. As recognized in the toxicological literature, such sub-lethal effects can be substantial. Murty (1986) stated that, "In the long run, these sub-lethal concentrations may prove more deleterious than the lethal concentrations, because subtle and small effects on the fish may alter their behavior, feeding habits, position in the school, reproductive success, etc." NMFS notes that Murty's observations regard individual fish responses, and that these must be extrapolated to population exposures that persist over time, and that will all cohorts of the affected populations.

As summarized by LaLiberte and Ewing (2006), laboratory experiments have shown that the sub-lethal concentrations of individual pollutants can impair physiological functions at every stage in the life history of salmonids:

1) They interfere with the biochemical machinery of the cells.
2) They show various neurotoxic effects that interfere with normal behavior.
3) They inhibit the olfactory system in such a way to interfere with homing, predator avoidance, and spawning.
4) They interfere with the immune system, leading to increased mortality from diseases.
5) They increase the incidence of carcinogenesis through oxidized metabolites, DNA adducts and interference with DNA repair mechanisms.
6) They interfere with developmental processes, leading to reduced fertility, increased mortality of the young, and teratogenesis.
7) They act as endocrine disruptors, causing interference with the intricate balance of hormones needed for reproduction, osmoregulation, and homeostasis.
8) PBTs released at any concentration level are very probably harmful to Chinook salmon, and other organisms, because of their persistent and bioaccumulating characteristics.

These changes in physiology, behavior, reproduction, and health among multiple individuals can in turn have significant effects on the population structure and number of Chinook salmon in the Puget Sound. The danger of these changes is that they are not detectable by current techniques in fisheries biology. The result is a decline in population numbers from causes that cannot be clearly identified.

Metals. Metals have a number of similar toxic effects on fish because of their similar properties. Most metals tend to accumulate in the gill tissue, where the metals form precipitates with the mucus. This leads to decreased ventilation, coughing responses, decreased oxygen and carbon dioxide exchange, and a depletion of energy reserves. The depletion of energy reserves causes decreased swimming ability and a slower response to predators (LaLiberte & Ewing 2006).

Metals tend to accumulate within the body of the fish by binding to phosphate and sulfide groups of various proteins. When the sulfhydryl groups of enzymes are bound, the enzyme activity can be inhibited potentially causing major disruption of physiological functions and a general decline in fish health (Leland & Kuwabara, 1985; Kime, 1998). At high enough concentrations, osmoregulatory and hormonal systems can cease to function (LaLiberte & Ewing 2006).

Some metals also interfere with olfaction in salmonids (Klaprat et al. 1992). Salmon use olfaction as the major sensory input describing the environment around them. Olfaction has been shown to play important roles in predator avoidance (Scholz et al. 2000; Brown & Smith 1997; Hiroven et al. 2000), recognition of kin (Quinn & Busack 1985; Olsen 1992), homing of adults to natal streams (Wisby & Hasler 1954; Hasler & Scholz 1983; Stabell 1992), and spawning rituals of adults (Sorensen 1992; Olsen & Liley 1993; Moore & Waring 1996).

Heavy metals also interfere with the workings of the immune system in salmonids (Anderson 1989) but the mechanism of interference is not clear (Kime 1998). Metals may affect the immune system directly or the response could result from a stress reaction that elevates cortisol which subsequently results in immunosuppression (Schreck 1996). Suppression of the immune system increases infection of salmonids to bacteria, fungi, viruses, and parasites. Such infections decrease the vitality of the fish and increase the chances of mortalities due to osmotic imbalance, inability to feed, or predation (LaLiberte & Ewing 2006).

NPDES permits of WWTP's implicitly allow for the discharge of persistent toxic chemicals, such as PAHs, some metals such as mercury, and PBDE's, because permit limits or treatment requirements are not specified in such permits. Potentially toxic constituents that typically sorb to suspended solids can settle out of the water column in, and beyond, the permitted mixing zone. In the absence of source controls or sufficient treatment, toxicant accumulation can occur in outfall mixing zone sediments and nearby vicinities, and these toxic contaminants remain perennially available to organisms for uptake and potential bioaccumulation.

Copper. The maximum measured effluent concentration of copper from the previous permit cycle was 44 micrograms (ug)/liter (L). This translates to a calculated maximum concentration at the edge of the acute and chronic mixing zones of 0.90 µg/L and 0.54 µg/L respectively. Copper can be detected in multiple forms in the aquatic environment. It can be dissolved, or bound to organic and inorganic materials either in suspension or in sediment. Dissolved copper is highly toxic to a broad range of aquatic species including algae, macrophytes, aquatic invertebrates, and fishes. More than three decades of experimental results have shown that the sensory systems of salmonids are particularly vulnerable to the neurotoxic effects of dissolved copper. Sensory system effects are generally among the more sensitive fish responses and underlie important behaviors involved in growth, reproduction, and (ultimately) survival (i.e., predator avoidance). Recent experiments on the sensory systems and corresponding behavior of juvenile salmonids show that dissolved copper directly damages the sensory capabilities of salmonids at low concentrations. These effects can manifest over a period of minutes to hours and can persist for weeks.

Effects of copper are difficult to clarify in the natural environment because of the wide variety of reactions that it undergoes with common waste stream components. Copper forms insoluble precipitates at low concentrations in the presence of a number of anions. Therefore, toxicity depends strongly on pH and hardness of the water used for experiments (Alabaster & Lloyd 1982; Sorensen 1991). Toxicity also depends upon temperature and dissolved oxygen in the water (Alabaster & Lloyd 1982; Lloyd 1961). Organic compounds such as humic acids and suspended solids can lower the toxicity of copper. These compounds are thought to act as ion-exchangers and preferentially bind aqueous copper (Brown et al. 1974).

A large body of scientific literature has shown that fish behaviors can be disrupted at concentrations of dissolved copper that are at or slightly above ambient concentrations (i.e., background). Analysis conducted by Hecht et al. (2007) predicted a substantial 24.2 percent reduction in olfaction at a dissolved copper concentration of 0.59 µg/L above background levels. Additionally, they calculated an acute criterion maximum concentration (CMC) using the Biotic Ligand Model (EPA 2007). The CMC is an estimate of the highest concentration of a substance in surface water to which an aquatic community can be exposed briefly without resulting in an unacceptable effect (EPA 2002). The EPA sets acute water quality criteria by calculating an acute CMC (Stephan et al. 1985). The estimated acute CMC using measured and estimated water quality parameters from Sandahl et al., (2007) was 0.63 µg/L with a range from 0.34 to 3.2 µg/L. This paper also presented examples of benchmark concentrations for juvenile salmonid olfactory function based on recent data. Benchmark concentrations of 0.18–2.1 µg/L (above background levels) corresponded to reductions in predator avoidance behavior of approximately 8 to 57 percent.

Baldwin et al., (2003) used electrical potential readings from the olfactory epithelium as a measure of the olfactory responsiveness of natural odorants in Chinook salmon exposed to copper. Copper inhibited the responsiveness of the epithelium to odorants within 10 minutes of exposure. Inhibitory responses occurred in a dose-dependent manner in a range of copper concentrations from 1.0 to 20.0 µg/L. Notably, inhibition was not dependent on hardness of the water.

Hansen et al., (1999) examined the avoidance of Chinook salmon and rainbow trout to water polluted with copper or cobalt. Chinook salmon were found to be the most sensitive, avoiding water containing as low as 0.7 ug copper/L. Rainbow trout avoided water containing as low as 1.6 ug copper/L. When fish were acclimated to water containing 2 ug copper/L, rainbow trout avoided water with concentrations of 4 ug copper/L and preferred clean water, but Chinook salmon did not avoid any concentration of copper and did not have a preference for clean water. The authors concluded that the Chinook salmon exposed to low levels of copper had their olfactory senses impaired to the point where they could not avoid water of lethal concentrations. This impairment also probably had deleterious effects on predator avoidance, homing, and spawning activities.

Copper has also been documented to cause immunosuppression in most species of fish and this can be particularly damaging in salmonids. Baker et al., (1983) showed that exposure of Chinook salmon and rainbow trout to sub-lethal concentrations of copper caused an increased susceptibility to infection by *Vibrio anguillarum*. Similarly, Hetrick et al., (1979) showed that exposure to copper increased susceptibility of rainbow trout to infectious hematopoietic necrosis virus and Knittel (1981) found that steelhead exposed to copper were more susceptible to *Yersinia ruckeri*. Anderson et al., (1989) showed that exposure of isolated rainbow trout spleen cells in vitro to copper caused inhibition of the antibody-producing cells (LaLiberte & Ewing 2006).

Dissolved copper's effect on salmonid olfaction in saltwater environments remains a recognized data gap and it is presently uncertain whether the BMC thresholds derived in (Hecht et al. 2007) apply to salt water environments. Estuarine and nearshore salt water environments, despite their higher salinity and hardness may or may not confer protection against dissolved copper-induced olfactory toxicity. One source of this uncertainty is whether or not free copper is the sole species of copper responsible for olfactory toxicity. In freshwater, evidence suggests that Cu2 is not the only toxic species that adversely affects olfaction in fish (McIntyre et al., in press) as well as more conventional endpoints such as mortality (Niyogi & Wood 2004). Other copper species (e.g., CuOH; Cu1+) will also bind to the gill, thereby causing toxicity (Niyogi & Wood 2004). While the physiological basis for salmonid olfaction is well characterized, the transition to saltwater may involve important changes in olfactory receptor neuron function that ultimately influence the expression of the as yet unidentified ligands for dissolved copper (Hecht et al. 2007).

Chlorine. Water quality based limits for total residual chlorine levels in the proposed permit have been established at 0.36 mg/L (360 µg/L) as the average monthly effluent concentration and 0.50 mg/L (500 µg/L) as the daily maximum effluent concentration. Within the past permit cycle, chlorine in the Solo Point effluent has been reported as high as 0.16 mg/L (160 µµg/L) for a daily average and as 0.80 mg/L (800 µg/L) for a daily maximum. Calculations in the permit fact sheet for this action demonstrate a reasonable potential for the discharge to violate Washington State Water Quality Criteria for chlorine (13 µg/L as a one hour average and 7.5 µg/L as a four day average). The highest projected concentrations are 25 µg/L (one hour criteria) at the acute mixing zone edge and 15 µg/L (four day criteria) at the chronic mixing zone edge.

Sprague and Drury (1969) found that rainbow trout were killed at 0.01mg/L (10 µg/L) in 12 days, and they avoided a concentration of 0.001 mg/L (1 µg/L). Fifty percent of brown trout were killed at 0.02 mg/L (20 µg/L) within 10.5 hours and at 0.01 mg/L (10 µg/L) within 43.5 hours (Pike 1971).

Chlorine is a powerful oxidizing agent with a high solubility in water (Brungs 1973). Chlorine in water may be present as free available chlorine in the form of hypochlorous acid or hypochlorite ion or both. Chlorine may also be present as combined available chlorine in the form of chloramines (mono-, di-, and tri-) and other chloro derivatives.

The toxicity to aquatic life of chlorine wastes depends not on the amount of chlorine added but on the concentration of residual chlorine remaining and on the relative amounts of free chlorine and chloramines (Brungs 1973). Doudoroff and Katz (1950) and Merkens (1958) stated that toxicity of free chlorine is apparently of the same order as that of chloramines, and a measure of residual chlorine is generally adequate to define chlorine toxicity.

Enslow (1932) reported not only that chlorination results in free chlorine and chloramines, but also that chlorination of many organic compounds closely allied to compounds present in wastewater effluent results in the production of end products entirely different from the original material. Allen et al., (1948) found that cyanogen chloride formed by the reaction of chlorine with thiocyanate was toxic to aquatic life. Potassium sulfocyanide was also converted to a more toxic material after chlorination.

Rosenberger (1971) using coho salmon, determined, as did Merkens (1958) that free chlorine is the most toxic form of chlorine and that dichloramine appears to be more toxic than monochloramine. He also concluded that larger fish will die faster than smaller fish because a large fish has less gill surface area per unit body than a small fish, and chlorine attacks the gill tissue. Rosenberger (1971) further stated that the toxicity curves for the chloramines differ from the curve for free chlorine in that the lethal effect of free chlorine is more rapid. In studies with chlorinated wastewater from treatment plant effluent, lethality was not as rapid at comparable concentrations of residual chlorine when no free chlorine was present (Arthur 1971). In most surface waters ammonia is present in amounts sufficient to allow no free chlorine after a brief time for reaction.

Holland et al., (1960) determined that dichloramine was more toxic than monochloramine and those chloramines were more toxic than chlorine to salmon in seawater. Those results disagree with Merkens (1958) and Rosenberger (1971) perhaps because the latter studies were conducted in fresh water, and the chemistry of chlorine may not be similar in both waters.

Several of the studies previously described indicated that salmonids were the most sensitive fish species to chlorine and its derivatives. Laboratory bioassays support this generalization. A residual chlorine concentration of 0.006 mg/L (6 µg/L) was lethal to trout fry in 2 days (Coventry et al 1935). Brungs (1973) concluded trout, salmon, and some fish-food organisms are more sensitive to chlorine than warmwater fish, snails, and crayfish. He also summarized that chronic toxicity effects of residual chlorine on growth and reproduction occur at much lower concentrations than acutely lethal concentrations and in areas receiving wastes treated continuously with chlorine. It was recommended that total residual chlorine should not exceed 0.01 mg/L (10 µg/L) for the protection of more resistant organisms only, or 0.002 mg/L (2 µg/L) for the protection of most aquatic organisms.

Stober et al., (1980) found that coho salmon were also highly susceptible to chlorine toxicity. Coho salmon did not survive an exposure of 7.5 minutes at 0.5 mg/L total residual oxidant (TRO) or more. Survival of 100 percent after 60 minutes of exposure occurred at TRO concentrations of less than 0.1 mg/L. Significant avoidance responses by juvenile coho salmon to chlorine concentrations in sea-water of 2, 10, 25, 50, 100 and 500 μg/L were observed.

Yearling coho salmon were exposed for 12 weeks to chlorinated sewage plant effluent diluted with seawater under continuous flow conditions by Buckley et al., (1976). Concentrations of effluent at 1.1 and 3.6 percent (9 and 30 μg/L of total residual chlorine, respectively) resulted in reductions of hemoglobin and hematocrit to levels indicative of anemia. Observations of the erythrocytes (red blood cells) revealed lysed and degenerating cells, increased numbers of circulating immature cells, and abnormal cells. The highest tested no-effect concentration of total residual chlorine was 3 μg/L in 0.3 percent effluent.

Mattice and Zittel (1976) plotted available data on median effect concentrations of total residual chlorine on freshwater and marine organisms and estimated acute and chronic toxicity thresholds for each group. These thresholds represent a dose-time combination below which there are either no deaths (acute threshold) or no effect, no matter how long the exposure (chronic threshold). The freshwater and marine acute thresholds are different, but both were time dependent. The chronic toxicity thresholds are constant for freshwater and marine organisms. As a result of apparent basic differences in sensitivity of freshwater and marine organisms, which may be the result of different total residual chlorine chemistry, the times at which the chronic thresholds begin are different. For freshwater organisms the chronic threshold begins at nearly 1,000 minutes of exposure, whereas for marine organisms it begins at just over 100 minutes. The acute toxicity and chronic toxicity threshold for effects from chlorine on saltwater organisms were estimated at concentrations of 30 and 20 μg/L respectively. They also found that marine organisms appeared more susceptible to acute doses of chlorine. Few freshwater organisms were affected by exposures of less than 10 minutes, while several marine species suffered adverse effects at even shorter durations. Conversely, the freshwater organisms appear more sensitive to chronic exposures at low concentrations.

Bis(2-ethylhexyl)phthalate. Phthalate esters (semi-volatile organic compounds) such bis(2-ethylhexyl)phthalate (DEHP) are toxic and harmful to aquatic organisms. They have been identified as being present in the current discharge with DEHP detected at a level of 6.22 μg/L. Sediment monitoring was conducted in 1995 at 14 sites near the main outfall and an additional site at a reference location in Carr Inlet. Chemical analysis and biological tests were performed on the sediment samples from the top two centimeters of sediment. The chemical concentrations for DEHP did not meet Sediment Management Standards in all but one sample, including the reference site. Zanotelli et al., (2009) found that this chemical significantly blocked growth in both male and female young guppy fish. Different concentrations of DEHP (0.1–10 μg/L) applied continuously showed significant effects in as little as 14 days from the start of exposure.

Polybrominated Diphenyl Ethers (PBDEs). PBDEs are members of a broad class of brominated chemicals used as flame retardants (WDOE and WDOH 2006). Polybrominated Diphenyl Ether levels have not been tested in the Solo Point WWTP effluent; however, as discussed previously, they are regularly found in Puget Sound waters and sediments. PBDEs are considered a chemical of concern by the Washington Department of Ecology and Puget Sound Partnership, and recognized sources include wastewater discharges, surface runoff, atmospheric deposition, oil spills and CSOs (Crowser et al. 2007). The family of PBDEs consists of 209 possible substances, which are called congeners. They have been added to plastics, upholstery fabrics and foams in common products like computers, TVs, furniture and carpet pads. There are three main types of PBDEs used in consumer products and each is made up of a mixture of different brominated diphenyl ether (BDE) congeners; Penta-BDE, Octa-BDE and Deca-BDE. Individually they have different uses and different toxicity. Lower brominated PBDEs average five or less bromine atoms per molecule and are regarded as being more dangerous. Higher brominated PBDEs average greater than five bromine atoms per molecule. The congeners BDE-47 (four bromine atoms) and BDE-99 (five bromine atoms), found in penta-BDE, are the most frequently detected and found in the highest concentrations in organisms (WDOE and WDOH 2006).

Studies indicate that PBDEs are found throughout the natural environment (air, soil and sediments), and are building up in animals throughout the food chain. PBDEs have been introduced into the marine environment by various processes, such as discharge of domestic sewage and industrial wastewater, agricultural inputs, runoff from nonpoint sources and atmospheric deposition (Alaee et al. 2003). Lema et al., (2008) found developmental disorders such as reduced growth, abnormal morphology, irregular cardiac function, and altered cerebrospinal fluid flow in zebrafish upon exposure to high concentrations of PBDEs (100-5000 μg/L). Brief exposure to PBDE 47 causes morphological abnormalities during development and growth of embryos in zebrafish. Chronic exposure to PBDE 47 can disrupt thyroid hormones and affect various key enzymes regulating the production of steroids and receptors in fish gonads. This alters the levels of hormones that stimulate the growth and activity of the gonads, which impairs fish reproduction (Muirhead et al. 2006).

Pharmaceuticals and Personal Care Products. Pharmaceuticals and personal care products (PPCPs) are an emerging environmental and human health issue and have been identified as constituents discharged into Puget Sound in a recent survey of effluent from five municipal wastewater treatment plants (Lubliner et al. 2010). PPCPs refer to any product used by individuals for personal, health or cosmetic reasons. They are present at low concentrations in surface water, groundwater, soils, sediments, marine waters, and drinking water. Researchers monitoring the environment find PPCPs virtually everywhere domestic wastewater is discharged. PPCPs enter the environment as they pass-through the human body or when unwanted PPCPs are disposed in the trash or down the drain. Other significant sources include livestock, aquaculture, pets, and agriculture. PPCPs have not previously been monitored in the Solo Point WWTP effluent.

There is considerable evidence that fishes inhabiting waters that receive effluent from municipal WWTPs are exposed to chemicals that effect reproductive endocrine function. Male fish downstream of some WWTP outfalls produce vitellogenin (egg yolk precursor protein) mRNA (messenger ribonucleic acid, which carries information from DNA in the nucleus to the ribosome sites of protein synthesis in the cell), and protein associated with oocyte (an immature ovum or egg cell) maturation in females, and early-stage eggs in their testes (Jobling et al. 1998).

This feminization has been linked to the presence of estrogenic substances such as natural estrogen, 17 beta-estradiol (E2) and synthetic estrogen, 17 alpha-ethenylestradiol (EE2). These substances are usually found in the aquatic environment at low parts per trillion concentrations, typically less than 5 nanograms (ng)/L (Zhou et al. 2007). Synthetic estrogen is used in birth control pills (EE2) and is one of the more potent estrogens and has been linked to the feminization of male fishes in rivers receiving municipal wastewater (Thorpe et al. 2003). Laboratory studies have shown decreased reproductive success of fish exposed to less than 1-5 ng/L of EE2 (Parrott & Blunt 2005).

Kidd et al., (2007) showed that chronic exposure of fathead minnows to low concentration (5-6 ng/L) of EE2 led to feminization of males through the production of vitellogenin mRNA and protein, impacts on gonadal development as evidenced by intersex in males and altered oogenesis (egg cell production) in females. This ultimately caused a near extinction of this fish species from the lake where they were being studied. This demonstrated that the concentrations of estrogens and their mimics observed in freshwaters can impact the sustainability of wild fish populations.

Conventional wastewater treatment systems do not do a good job of removing or destroying PPCPs. No single treatment process will completely remove all the trace organic chemicals to less than detection levels. There are thousands of chemicals used in PPCPs. Their presence in the environment depends upon their individual chemical structure and the frequency of their use. The occurrence and concentrations and of PPCPs are correlated to effluent dominated water.

The treatment processes which have the highest removal efficiencies include ozonation, nanofiltration, granular activated carbon, and reverse osmosis. Some researchers have concluded that a multi-barrier approach, using multiple treatment processes is the most effective. Reclaimed water provides a higher level of treatment beyond conventional wastewater treatment plants, and subsequently produces water which is lower in PPCP concentrations.

Ammonia. The acute water quality criterion for this discharge is 10.64 mg/L and the chronic criteria is 1.6 mg/L at this facility. The ammonia 95th percentile value reported on the monthly discharge monitoring reports was 4.67 mg/L, and the mean was 2.86 mg/L. Ammonia dissolves in water and may directly exert a toxic effect on organisms in Puget Sound waters. Several studies have documented negative changes in behavior that occur at sub-lethal concentrations of un-ionized ammonia, beginning at 0.05 mg/L (Woltering et al. 1978). Changes in gill permeability occurred at concentrations of non ionized ammonia as low as 0.09 mg/L (Lloyd & Orr 1969). Because salt and water regulation in estuarine fish occurs at the gill surface, changes in the gill permeability can reduce the ability of fish to survive. These sub-lethal concentrations of ammonia can cause malformation of trout embryos and histopathological changes (i.e., tissue changes characteristic of disease) in gills, kidneys, and livers of fish (Flis 1968; Smith & Piper 1972; Thurston et al. 1978; Soderberg 1985; EPA 1986; Soderberg 1995). Salmonids that are exposed to these concentrations of ammonia reduce their feeding and thereby reduce their growth and survival (Soderberg 1995).

Bioaccumulation in Prey Species. Some contaminants that are likely to be present in the Solo Point wastewater effluent are hydrophobic and lipophilic (fat loving) and can bioaccumulate in listed species. Bioaccumulation is the net result of contaminant uptake anddistribution in animals from water, sediment, and dietary exposure. Consequently, listed salmonids will have indirect exposure to contaminants when they consume food organisms that have accumulated the pollutants (McCain et al. 1990). The lipophilic compounds found in the food sources tend to bioaccumulate in the lipid-rich tissues of the salmon. These concentrated pollutants can have consequences on all aspects of the biology of the salmon and may affect the survival of populations (Arkoosh et al. 1998).

The lipid solubility of hydrophobic contaminants facilitates their bioaccumulation in fish and the biomagnification of these chemicals in higher order animals that may consume them such as killer whales, as discussed. Biomagnification is defined as the increase in concentration of pollutants as they move to higher and higher trophic levels. Biomagnification has been demonstrated convincingly only for a few chemicals, such as methylmercury (Bargagli et al. 1998), PCBs (Oliver & Niimi 1988; Evans et al. 1991), and some dioxins and furans (Opperhuizen & Sijm 1990; Sijm et al. 1993). Of the effluent constituents identified, a subset are known to have bioaccumulative tendencies. These include some metals, the phthalates, polybrominated diphenyl ethers (PBDEs), polycyclic aromatic hydrocarbons (PAHs) and (PCHs).

PBDEs have been detected in peregrine falcon eggs, killer whales, harbor seals, fish, and in polar bears in the Arctic, which indicates that these chemicals can move great distances from where they are made and used. Once in the environment, PBDEs can last a long time depending on surrounding conditions such as the availability of water, organic compounds or sunlight. PBDEs, especially those with higher numbers of bromines such as deca-BDE, can break down into lower brominated PBDEs, which are more bioaccumulative. Pathways for PBDEs from products to the environment are not well understood (WDOE and WDOH 2006).

In its original form Deca-BDE is considered to be relatively safe. But, laboratory studies indicate that Deca-BDE breaks down through exposure to sunlight and biological activity into more toxic forms. Degradation products include other PBDEs such as the lower brominated congeners found in Penta-BDE. These substances have been proven to have a greater environmental impact and are known to bioaccumulate, biomagnify and have greater toxicity. Therefore, the Deca-BDE that is already in the environment is likely to be a long-term source of the more toxic forms of PBDEs long into the future (WDOE and WDOH, 2006). Manufacturers of Penta- and Octa-BDE in the U.S. agreed to voluntarily stop producing these two forms of PBDEs at the end of 2004. With the discontinuation of Penta- and Octa-BDE, Deca-BDE now accounts for 100 percent of PBDE usage (WDOE and WDOH 2006).

Marine organisms in higher trophic levels (e.g., fish) can concentrate PBDEs from water or their diet. Oral exposure of PBDE 47 delayed hatching and reduced fecundity in the Japanese medaka and fathead minnows (Muirhead et al. 2006). These researchers also found that results from both the medaka and fathead minnow feeding studies indicate that PBDE-47 is well absorbed from the fish gastrointestinal tract. The relatively slow decline in the medaka PBDE-47 body levels and correspondingly long biological half-life are indicative of the limited capacity of fish to excrete PBDE-47. Combined, these properties (efficient uptake and slow elimination) explain the tendency of PBDE-47 to bioaccumulate to significant levels in fish (WDOEand WDOH 2006).

Lam et al., (2010) found that PBDEs altered larval settlement of marine benthic polychaetes. They concluded that their study clearly demonstrated that environmentally realistic concentrations of PBDE 47 in sediment can affect polychaete settlement in species specific and dose-dependent manners. Because sustainability of marine benthic polychaete populations is highly dependent upon larval recruitment, alteration in settlement patterns of different species may change the normal recruitment patterns and eventually species composition of the benthic community. Polychaetes are a documented food source (Duffy et al. 2010) for young juvenile Chinook salmon in nearshore areas, thus bioaccumulation of this compound is highly likely.

The following issues arise when assessing the risks of lipophilic contaminants that have little or no affinity for water:

1) Delivery to the fish. Contaminants that are poorly soluble in water and must be delivered to the fish either through very low water concentrations, through sediments, or indirectly through the food supply (Spacie & Hamelink 1985).

2) Bioaccumulation occurs in different tissues at different rates. For a particular assay, it can be difficult to determine which tissue is influencing the results.

3) Non-equilibrium conditions are present for long periods of time. It is difficult to determine the concentration of toxicant that is producing the results obtained.

4) Individual fish may receive different "dosages" either from different feeding habits, different resting areas in the exposure area, or differences in physiology. A correlative approach using individual fish is required for analysis.

5) Fish have the ability to biotransform PAHs and PCHs so that measured quantities may not reflect the concentrations that initiated the results obtained.

It is therefore difficult to relate tissue concentrations of some toxicants to physiological effects that may be attributed to them. Studies of hydrophobic contaminants such as PBDEs, PCBs and PAHs have to rely on tissue concentrations for developing relationships with deleterious changes to fish populations. Concentrations in sediments or in the water are not necessarily related to the amounts of pollutants to which the fish tissues are exposed (LaLiberte & Ewing 2006).

Decreased Levels of Dissolved Oxygen. Organic materials released to the environment from WWTP effluent and other sources undergo oxidative metabolism by bacteria. This oxidative capacity is measured crudely as a process known as biological oxygen demand (BOD). If enough organic material is released into the environment, the oxygen concentrations in the water can decrease to levels that cause respiratory distress, lack of feeding and growth, and death in salmon (Davis 1975; Kramer 1987).

Eutrophication of a waterbody can also occur from organic enrichment. Eutrophication refers to the process in which elevated nutrient levels result in excessive primary production by the plant community, potentially leading to a reduction in dissolved oxygen concentrations as increasing amounts of organic material are produced and decomposed within the water column. According to the South Puget Sound Water Quality Study, Phase 1 (Albertson et al. 2002), eutrophication will likely have the greatest impact in South Sound areas where flushing is low, where strong density stratification occurs, and where phytoplankton growth may be nutrient limited, such as in bays and inlets. Increases in nutrient (nitrogen and phosphorous) loads can accelerate the eutrophication process.

Albertson et al. (2002) identified low concentrations of dissolved oxygen in bottom-water samples collected during late summer, reaching the threshold of biological stress (5 mg/L) in Carr and Case inlets and exceeding the threshold in Budd Inlet (2 mg/L). Phytoplankton productivity can be limited by nitrogen availability during the growing season. Nutrient addition substantially enhanced measured rates of primary production, especially in late summer. Sensitivity to oxygen demand, settling rates, and algal metabolism rates show that dissolved oxygen is affected more by nutrient-driven processes than by direct biochemical oxygen demand (BOD) loading.

In a September 2007 South Puget Sound study of dissolved oxygen, the WDOE found that wastewater treatment plants contributed 80 percent of the watershed dissolved inorganic nitrogen load to South Puget Sound (WDOE 2008a). Nitrogen loads in late summer are particularly important because this is the period during which dissolved oxygen levels are generally the lowest. Direct point source inflow volumes represent only 2 percent of the watershed inflows to South Puget Sound. Yet, they also represent 36 percent of the total nitrogen load and 54 percent of the total phosphorus load, as well as 43 percent of the organic nitrogen load and 30 percent of the dissolved inorganic nitrogen load. There is no proposed effluent limitation for nitrogen or phosphorus in the new permit for the Solo Point WWTP outfall.

Anticipated Exposure and Response of Puget Sound Fishes to Solo Point WWTP Effluent. The WWTP outfall (last diffuser port) is located 500 feet off shore about 70 feet below the elevation mean of the lower of the daily low waters (MLLW). It is assumed that the distance from shore is measured from the riprap shoreline. Since the outfall is actually a series of 14 ports that encompass a distance of 130 feet, the first diffuser port would be located 370 feet from the shoreline. When accounting for an additional 270 feet at each end of the diffuser, the chronic mixing zone would begin at a point 100 feet from shore and extend a total distance of 770 feet from shore. Given the angle at which the outfall pipe extends relative to the shoreline, a corner of the chronic mixing zone may actually be within 35 feet of the shoreline. Assuming a constant slope of the bottom, the chronic mixing zone will encompass depths from approximately 5-108 feet. The acute mixing zone is one tenth of the chronic mixing zone and is closely associated with diffuser ports (23 feet on either side for its length and 27 feet out from each end).

*Puget Sound Chinook Salmon***.** Adult and juvenile Chinook salmon as well as adult and juvenile steelhead use the area adjacent to the Solo Point WWTP outfall. However, for this analysis, NMFS considers the associated nearshore environment (30 meters in depth and shallower) to be the portion of the action area predominantly used by juvenile Chinook salmon and steelhead.

Duffy (2003) studied the early marine distribution and trophic interactions of juvenile salmon at various locations in northern and Southern Puget Sound (south of the Tacoma Narrows) Sampling was performed from April through September in 2001 and 2002. Spatial and temporal differences in distribution, size structure, and diet among species (chum, pink, coho, and Chinook salmon) of salmon and between hatchery and unmarked (coho and Chinook salmon) salmon were evaluated. Sampling was conducted with a floating beach seine and net trawls. Specific sampling locations for beach seining in South Puget Sound (Figure 15) are as follows: 1) Wollochet Bay (southern end of bay mouth/Hale Passage); 2) Sunset Beach (along railroad line south of Day Island); 3) Chambers Creek (north of creek mouth); 4) Gordon Point (public beach at Steilacoom); 5) Solo Point (south of Solo Point boat launch); and 6) Solo Point Creek (mouth of small creek, overflow pipe). As part of the study, coded wire tags were recovered from marked hatchery fish and read by WDFW (L. Anderson, WDFW, unpublished data), see Table 10.

Duffy (2003) study focused on two major sampling areas: a northern Puget Sound region encompassing Possession Sound/Port Susan/Port Gardner in the Whidbey basin, and a southern Puget Sound (SPS) region, south of the Tacoma Narrows sill. Unfortunately, data from these sampling efforts were combined by Duffy and analyzed based upon these two sampling areas and not by specific sampling locations. Notwithstanding, there is significant overlap of the southern Puget Sound study region, and the data associated with the action area identified in this Opinion. Therefore, NMFS assumed an equivalent proportion of natural origin Chinook salmon would be found within the action area as was identified in Duffy (2003) for hatchery origin Chinook salmon. It is also assumed that natural predation encountered by stocks sampled in the marine zones was inherent within the study and therefore will not be considered as part of this analysis.

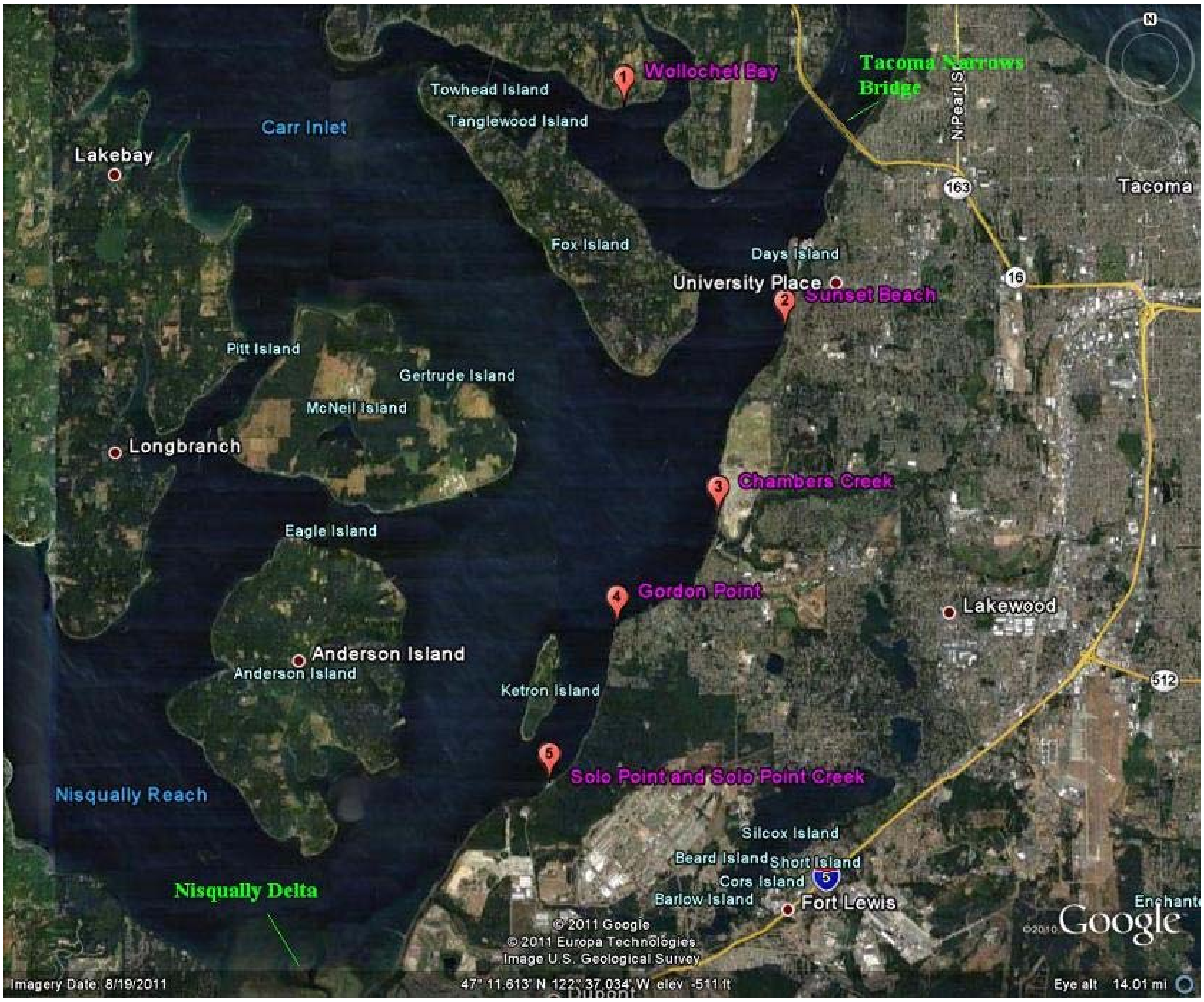

Figure 15: Sampling Site Locations of Puget Sound Chinook by Duffy (2003)

Table 10: South Puget Sound Coded Wire Tag Recoveries of PS Chinook

River Origin (Population)	Number Recovered	Percent (approximate)
Nisqually River	34	46
Puyallup River	17	24
Duwamish/Green River	11	15
Snohomish River	8	11
East Kitsap, South of Tacoma Narrows	1	1
East Kitsap, North of Tacoma Narrows	1	1
Skagit River	1	1
Skokomish/Dosewallips Rivers	1	1
Total	74	100

In Table 11, the Geometric mean of natural origin Chinook salmon spawners was taken from Table 2 in NMFS (2010a). The estimated smolt recruitment was calculated using the formula in Brakensiek (2006) which is: the number of spawners (S) multiplied by juvenile fish productivity (Jpro), divided by one plus the number of spawners (S) multiplied by juvenile fish productivity (Jpro), divided by juvenile fish carrying capacity (Jcap) or *smolt recruitment = (S * Jprod) / 1 + (S * Jprod) / Jcap)*.

Table 11: Estimated Puget Sound Chinook Salmon Natural Origin Smolt Recruitment and Natural Origin Adult Spawning Escapement

Population	Smolt Recruitment[1]	Geometric Mean of Natural Origin Spawners[2]
Skagit River (includes Upper Skagit, Lower Skagit, Upper Sauk, Lower Sauk, Suiattle and Cascade populations)	4,648,611	14,539
Snohomish River (includes Skykomish and Snoqualmie Populations)	1,081,605	4,309
Duwamish/Green River	760,500	3,615
White River	92,042	987
Puyallup River	141,456	969
Nisqually River	314,034	1,549

[1] Brakensiek (2006)
[2] NMFS (2010a)

In Table 12, numbers of fish (smolt & 0+) using South Puget Sound were determined by multiplying smolt abundance in Table 11 by the percent of South Puget Sound coded wire tag recoveries in Table 10 for each population. Not all coded wire tagged fish would be expected to encounter the chronic mixing zone associated with the WWTP outfall. In Duffy (2003) there was no indication as to the number of coded wire tag recoveries at each individual sampling site. To refine the number of fish exposed to the mixing zone, NMFS assumed an even distribution of tag recoveries at all sampling locations (17 percent). However, it is unlikely any fish sampled at the Wollochet Bay location would be exposed to the mixing zone so this proportion of the total Chinook population estimated to occur in South Puget Sound from tag recoveries (17%) was not considered likely to be exposed and is not included in the total estimate of the number of fish that would encounter the mixing zone. Therefore, the number of fish (northern populations traveling south through the Tacoma Narrows) using South Puget Sound was reduced by 17 percent to estimate total number of fish that are expected to be present in the action area (Table 12).

Table 12. Puget Sound Chinook Salmon Populations in South Puget Sound

Population	Nisqually River	Puyallup River	White River	Green/ Duwamish River	Snohomish River (Snoqualmie and Skykomish)	Skagit River (Upper Skagit, Lower Skagit, Upper Sauk, Lower Sauk, Suiattle, and Cascade)
Smolt Abundance[1]	314,034	141,456	92,042	760,500	1,081,605	4,648,611
Spawners[2]	1,549	969	987	3,615	4,309	14,539
Adult Equivalents (SA/S)	203	146	93	210	248*	169*
Percent Fish Using South Puget Sound[3]	100%	24%	24%	15%	11%	1%
Fish (smolt & 0+) using South Puget Sound	314,034	33,949	22,090	114,075	118,977	46,486
Total Fish (smolt & 0+) using South Puget Sound (Percent Fish Using South Sound minus 17%)	314,034	28,178	18,335	94,682	98,751	38,583

* Average of Populations
[1]Brakensiek (2006)
[2] NMFS (2010a)
[3]Duffy (2003)

To further estimate the numbers of fish likely to be exposed, NMFS assumes that fish traveling in a southern direction through the Tacoma Narrows will have a higher probability of utilizing nearshore areas closer to that natural constriction. As fish from northern populations (rivers) disperse into South Puget Sound the density of these individuals gradually diminishes as they move in a southerly direction, as reflected in the tag recoveries of Duffy (Table 10). Therefore a further reduction in fish numbers (using the nearshore) of five percent between the Duffy (2003) south sound sampling stations was conservatively assumed to estimate the number of fish that may actually encounter the chronic mixing zone and be exposed to discharged constituents.

The Puyallup River enters Puget Sound immediately north and east of the Tacoma Narrows. Considering this proximity, it is assumed there will be an uneven distribution of juvenile fish as they emerge from the narrows while traveling south. The NMFS assumes that approximately 50 percent will make their way along the east shoreline to the Sunset Beach sampling location, 40 percent along the west shoreline to the Wollochet Bay sampling sites and 10 percent traveling beyond the nearshore zone. Juvenile Chinook salmon making their way south through the narrows from rivers further to the north (Duwamish/Green, Snohomish and Skagit) are assumed to have an even distribution (45 percent west shoreline, 45 percent east shoreline and 10 percent out beyond the nearshore). So, by the time they make their way from the Sunset Beach sampling location to the chronic mixing zone the percentages will be reduced by another 15 percent (Table 13). This same distribution is assumed for juvenile Chinook salmon (45 percent moving west, 45 percent traveling north along the east shoreline and 10 percent out beyond the nearshore) as they disperse from the Nisqually River (Table 13). It is assumed that dispersal of juveniles from the Nisqually River (moving north) is similar to other populations as they move south. As a result, their numbers would be reduced by an additional 5 percent prior to encountering the mixing zone (40 percent).

Table 13: Yearly Puget Sound Chinook Salmon Estimated Exposure and Take

Population	Nisqually River	Puyallup River	White River	Green/ Duwamish River	Snohomish River (combined stocks)	Skagit River (combined stocks)	Fish Taken Smolts/Adults	
Total Fish (smolt & 0+) Using South Puget Sound (Table 12)	314,034	28,178	18,335	94,682	98,751	35,583		
Percent Using Nearshore	40%	35%	35%	30%	30%	30%		
Fish (smolt & 0+) Using Nearshore	125,614	9,862	6,417	28,405	29,625	10,675		
20% Exposure to Mixing Zones	25,123	1,972	1,283	5,681	5,925	2,135		
Take (5%) of Fish (smolt & 0+) from Acute Mixing Zone	1,256	99	64	284	296	107	2,106	
5% Adult Equivilents	6	1	1	1	1	1		11
Take (2.5%) of Fish (smolt & 0+) from beyond the Acute Mixing Zone	628	49	32	142	148	53	1,053	
2.5% Adult Equivalents	3	0	0	1	1	0		5
Total (7.5%) Take of Fish (smolt & 0+) from Effluent	1,884	148	96	426	444	160	3,159	
7.5% Adult Equivilents	9	1	1	2	2	1		16
80% Avoidance	100,491	7,890	5,134	22,724	23,700	8,540		
Forced to Deeper Water (25%)	25,123	1,972	1,283	5,681	5,925	2,135		

10%Predator Loss[1,2]	2,512	197	128	568	593	213	4,212	
10% Adult Equivilents	12	1	1	3	2	1		21
Total Smolt & 0+ Fish Taken	4,396	345	225	994	1,037	374	7,371	
Cumulative Adults Taken	22	2	2	5	4	2		38
Percent of Stock[3]	1.40%	0.24%	0.24%	0.13%	0.10%	0.015%		
Percent of ESU								0.14%

[1]Willette (2001)
[2]Matthews and Buckley (1976)
[3]Brakensiek (2006)

As reflected in Table 13, when fish encounter the mixing zone it is assumed that the majority (80 percent) will avoid it. This assumption is based upon Stober et al., (1980) where coho salmon consistently showed significant avoidance response to all test concentrations of chlorine in seawater. An implication of juvenile Chinook salmon avoiding the WWTP outfall plume is that some of them will try and swim around the chronic mixing zone. This behavior will cause a portion of the exposed fish to utilize deeper water habitat. Biologists typically assume that early/small Chinook salmon smolts and early outmigrating fry utilize the shallow nearshore to avoid predation by piscivorous predators, including the staghorn sculpin and larger salmon. Willette (2001) reports findings that support this theory. It was found that juvenile pink salmon in Prince William Sound leave the shallow nearshore when the biomass of large copepods, their food base, declined. With the juvenile pink salmon foraging in deeper water, the mean daily individual predator consumption of salmon increased by a factor of five. In the absence of conclusive studies on the increase of predation risk on juvenile Chinook salmon associated with WWTP outfalls in Puget Sound, the results from the Willette (2001) study leads NMFS to believe that an increase of predation on juvenile Chinook salmon as a result of modified migration and schooling behavior (e.g. swimming around the chronic mixing zone) is likely (Table 13).

In this case, it is also assumed that 25 percent of the fish trying to avoid the mixing zone will be forced to deeper water and exposed to increased predation. A mortality rate in saltwater of two percent a day (Matthews & Buckley 1976) was multiplied by five (Willette 2001) in arriving at a 10 percent predation loss for Chinook salmon that are diverted to deeper water upon encountering the mixing zone (Table 13).

It is assumed that the remaining 20 percent of fish in the nearshore zone will swim into and make some use of the chronic and acute mixing zones. Of the fish that are exposed to effects within the acute mixing zone, it is assumed five percent will suffer deleterious effects from the effluent (time, concentration and/or synergistic effects of constituents known to exist and those which are not tested for) in a sufficient quantity to eventually cause take in the form of harm or mortality (e.g., impaired predator avoidance, reduced disease resistance, reduced growth, etc.). This assumption is based upon concentrations of constituents in the effluent that will be encountered within the acute mixing zone, which makes up 10 percent of the chronic mixing zone. An additional 2.5 percent mortality was added to account for bioaccumulation/chronic effects beyond the acute mixing zone, for a total of seven and one half percent (Table 13). This is a reasonably conservative estimate based upon the body of information previously discussed on chronic bioaccumulative effects of some of the contaminants known and expected to be discharged through the effluent, and is akin to the application of a safety factor typically applied in conducting ecological risk assessments where exposure to aquatic contaminants is considered.

Puget Sound Steelhead. For purposes of estimating the exposure and response of Puget Sound steelhead to the mixing zone, it is assumed that the population of Puget Sound steelhead mainly exposed and affected by this project are from the Nisqually River. Steelhead smolt abundance from the Nisqually River was calculated based on the number of natural origin adult spawners from 2005 to 2009 (Ford et al. 2010) and a median value of smolt to adult return rate (0.35 percent for hatchery fish) from 1997 to 2002 (WDFW 2008). It is assumed that smolt to adult return rates will be the same for hatchery and natural origin fish. All Nisqually River steelhead make use of South Puget Sound (Table 14).

Table 14: Yearly Steelhead Numbers for the Nisqually River Winter Run Stock

Stock	Nisqually River
Smolt Abundance (SA)[1]	112,114
Spawners (S)[2]	402
Adult Equivalents (SA/S)	279
Percent Fish Using South Puget Sound	100 percent
Fish (smolt & 0+) using South Puget Sound	112,114

[1] WDFW (2008)
[2] Ford et al. (2010)

Juvenile steelhead migrating out of the Nisqually River generally make more use of deeper water and do not rely as heavily on nearshore areas. They are also larger (majority of smolts spend two winters in freshwater) and travel faster on their way out to the ocean (WDFW 2008). Therefore, no predator loss from avoidance of the mixing zone was assumed. Beach seining data collected by Fresh et al. (1979) in 1978 at the Dupont Dock and Tatsolo Point (both locations are within the action area identified in this Opinion) indicated steelhead made up one-tenth of one percent of the total catch of all salmonids captured (a total of only 8 steelhead were captured in 68 beach seines between February and July).

The analysis conducting during consultation assumes that the steelhead captured by Fresh et al. (1979) were of Nisqually River origin, as this source population is closest and largest in number to the action area. As steelhead outmigrants move rapidly to deep water upon emigration, steelhead from other populations are not expected to experience significant exposure because they would not be present in the nearshore of the action area. In support of this presumption, it is notable that no steelhead were captured from surface tow net sampling by Fresh et al. (1979) in the same sampling efforts at these locations. The analysis assumes the proportion of steelhead captured in the total catch of the Fresh et al. (1979) nearshore sampling effort would be representative of the proportion of the Nisqually smolt population that would be exposed, and applied a safety factor of 100 in consideration that the sampling conducted, while canvassing a broad temporal period, undoubtedly missed a proportion of outmigrant steelhead that would encounter the mixing zone.

As smolt outmigrant numbers were not available for 1978, we applied the median outmigrant numbers as identified in Table 14 (from 2005 to 2009) to derive the estimated number of individuals exposed. This presumption is recognizably conservative as well because smolt emigration in 1978 was likely greater than the median estimated from the latest status review. Thus, the 8 steelhead captured by Fresh et al. (1979) would represent 0.00714 percent of median smolt emigration population from 2005 to 2009. Applying the safety factor of 100, we assume that 800 steelhead smolts would be exposed to the effluent, or 0.714% of the median smolt emigration population. Because the fish that are exposed to those effects, it was also assumed that five percent of the exposed population would experience adverse effects from the effluent exposure to cause take, or 40 steelhead smolts. This assumption is based upon concentrations of constituents in the effluent that will be encountered within the acute mixing zone, which makes up 10 percent of the chronic mixing zone.

An additional two and one half percent mortality was added to account for potential bioaccumulation/chronic effects beyond the acute mixing zone, the many constituents which are not tested for in the effluent and any synergistic effects from the combination of chemicals, for a total of seven and one half percent, or 60 smolts (Table 15). Applying this 'bioaccumulation safety factor' is a reasonably conservative estimate based upon the body of information previously discussed on chronic bioaccumulative effects of some of the contaminants known and expected to be discharged through the effluent, and is akin to (but less than) the application of a safety factor typically applied in conducting ecological risk assessments where exposure to aquatic contaminants is considered. Considering the above assumptions, a total of 60 smolts, equivalent to 0.215 adults are considered to be taken by the outfall discharge during each year of operation.

Table 15: Yearly Steelhead Estimated Exposure and Take

Puget Sound Steelhead (winter run)	Fish (smolt & 0+) using South Puget Sound	0.7% Exposure to Mixing Zones *	Take (5%) of Fish (smolt & 0+) from Acute Mixing Zone	Take (2,5%) of Fish (smolt & 0+) from beyond the Acute Mixing Zone	Total (7.5%) Take of Fish (smolt & 0+) from Effluent	Adult Equivalents (smolt/ spawner)	Total Adults Taken Per Year	Percent of Spawning Stock	Percent of DPS
Population									
Nisqually River	112,114	800	40	20	60	279	0.215	0.05%	0.001%

*Assumes 100 fold safety factor from nominal capture results of Fresh et al. 1979

Yelloweye Rockfish, Canary Rockfish, and Bocaccio

For the purpose of this analysis, we refer to larval and pelagic juvenile rockfish as "larvae" because there is no clear delineation between these life stages, and each would be similarly affected from exposures within the proposed action. The effects on larval rockfish would occur chronically to each generation of larval lifestage over the five years in which the proposed action would occur. Larval rockfish have been documented near the outfall location (Waldron 1972). In addition, some larvae and pelagic juveniles of ESA-listed rockfish broadly disperse from the area of their birth (NMFS 2003; Drake et al. 2010) and are likely to be using habitat in the action area.

To estimate the amount of larvae exposed by this action on an annual basis (Table 16), NMFS: (1) estimated the mean abundance level of larval yelloweye rockfish, canary rockfish, and bocaccio based on data reported by Weis (2004) and recent recreational rockfish catch data compiled by WDFW (WDFW unpublished data); (2) estimated volume of the water column encompassed by the mixing zone boundaries; (3) calculated the mean number of larvae within the affected water column in the mixing zone; (4) assumed the density of larvae in the action area remained constant throughout the year and (5) assumed how many times the mixing zone volume would replace itself per year (Table 16).

The following analysis is based upon the methodology used in an Endangered Species Act Section 7 Formal Consultation for the Continued Use of Puget Sound Dredged Disposal Analysis Program Dredged Material Disposal Sites in Puget Sound (NMFS 2010c). Larval rockfish densities have been documented to range from approximately 0.65 to 2.6 (mean value of 1.625) fish per 35,315 cubic feet (1,000 cubic meters) in the San Juan Basin (Weis 2004). Rockfish larvae are difficult to identify from morphological features alone until they are several weeks to months old (Love et al. 2002), thus Weis did not identify species. To estimate the abundance of rockfish larvae in the water column while effluent is being discharged, the densities of all rockfish larvae reported in Weis (2004) were used, and bounded by the proportion of yelloweye rockfish, canary rockfish, and bocaccio caught in recent fisheries (WDFW 2010). For the purposes of this analysis, NMFS assumes the proportion of ESA-listed rockfish caught by recreational anglers compared to the total rockfish caught roughly represents the proportion of larval rockfish. The NMFS also assumes the range of densities for rockfish larvae caught in the San Juan basin (Weis, 2004) would be the same in the action area for this project, including those in South Puget Sound. This will result in a gross over-estimate of larvae affected because there are more rockfish in the San Juan region than the Puget Sound proper (Palsson et al. 2009). That in turn means there are likely more larvae in that region as well. Additionally, we assume that the distribution of larvae is uniform throughout the DPSs.

The proportion of adult yelloweye rockfish caught by recreational anglers from 2004 to 2008, as a proportion of the total rockfish catch, was 0.008 percent (WDFW 2010). Canary rockfish were 0.012 percent of the catch, and bocaccio were 0.00026 percent of the total rockfish caught (WDFW unpublished data). Multiplying the percentage of the recreational catch by the mean larval density taken from Weis (2004) estimated densities of yelloweye rockfish, canary rockfish, and bocaccio larvae at the Solo Point outfall can be derived. This calculation results in estimated densities of 0.014 yelloweye rockfish larvae, 0.021 canary rockfish larvae, and 0.000455 bocaccio larvae per 35,315 cubic feet (1,000 cubic meters). These estimated densities can then be used to approximate the number of larval ESA-listed rockfish that would be affected by the project within the mixing zone at any given time. Again, this estimate is likely much higher than would occur at the sites in Puget Sound proper because they are based on rockfish larvae density from the San Juan Basin (Weis 2004). There are more rockfish in the San Juan region than the Puget Sound proper (Palsson et al. 2009), thus there are likely more larvae in that region as well.

Table 16: Yearly Rockfish Estimated Exposure and Take

Puget Sound Rockfish	Mean Larval Rockfish (all spp) Density per 35,315 cubic feet[1]	Percent Caught by Anglers, 2004-2008[2]	Mean Larval Rockfish (ESA spp) Density per 35,315 cubic feet	Number of Volumes (35,315) in the Mixing Zone	Number of Larvae in the Mixing Zones	Replacement Volumes per Year	Total Number of Larva in the Mixing Zones Exposed per Year	Take (5%) of Larvae in the Acute Mixing Zone	Take (2.5%) of Larvae beyond the Acute Mixing Zone	Total (7.5%) Take of Larvae from Effluent	Yearly Production of Larvae (millions) per Adult Cohort[3]	Total Take (range) of One Adult Cohort
Yelloweye	1.625	0.008	0.013	410	5.3	24,480	130,478	6,524	3,262	9,786	1.2 - 2.7	.36 - .82%
Canary	1.625	0.012	0.0195	410	8.0	24,480	195,718	9,786	4,893	14,679	0.26 - 1.9	.77 - 5.8%
Bocaccio	1.625	0.00026	0.000423	410	0.2	24,480	4,241	212	106	318	0.02 - 2.3	.01 - 1.6%

[1]Weis (2004)
[2]WDFW (2010)
[3]Love (2002)

The volume of the mixing zone was calculated based on the length and width measurements provided in the BE (670 x 460 = 308,200), and estimated median depth across this area. As referenced in the BE, the depth at the diffuser is listed as 70 feet at a distance of 500 feet from shore. Assuming a constant slope, the median depth of the mixing zone would be 47 feet. Therefore, the chronic mixing zone volume is estimated at 14,485,400 (47 x 308,200) cubic feet. Dividing the mixing zone volume (14,485,400) by the area of mean larval rockfish density (35,314) produces a multiplication factor (410) for determining number of larvae in the mixing zone. Multiplying the mean densities of rockfish larvae by 410 produces an estimated total number of larvae expected to be present at any given moment within the mixing zone.

For this analysis it is assumed rockfish larvae are present throughout the water column at the above calculated densities in the action area and mixing zone on a year-round basis (Moser and Boehert 1991; Weis 2004). An average current speed of 1,320 feet per hour was utilized in determining how many times the mixing zone volume (and associated larvae) replaced itself during a 6 hour tidal cycle (1,320 x 6 = 7,920). The width of the mixing zone is 460 feet divided into 7,920 approximately equals 17 replacement volumes every six hours, or 68 (4 x 17) per day or 24,480 (68 x 360) per year. The current speed was considered to be constant throughout the mixing zone length, depth and in each direction (ebb and flood). It was assumed that all new larvae (no re-exposure) would be exposed during each ebb and flood tide. Numbers of larvae within the chronic mixing zone are multiplied by the number of replacement volumes in a year to determine the total larvae exposed per year.

Of the larvae that are exposed to effects from the acute mixing zone, it is assumed five percent will suffer deleterious effects from the effluent (time and concentration) in a sufficient quantity to eventually be lethal. This assumption is based upon concentrations of constituents in the effluent that will be encountered within the acute mixing zone, which makes up 10 percent of the chronic mixing zone. An additional two and one half percent mortality was added to account for potential bioaccumulative/chronic effects beyond the acute mixing zone, the many constituents which are not tested for in the effluent and any synergistic effects from the combination of chemicals, for a total of seven and one half percent (Table 16). Applying this 'bioaccumulation safety factor' is a reasonably conservative estimate based upon the body of information previously discussed on chronic bioaccumulative effects of some of the contaminants known and expected to be discharged through the effluent, as previously discussed in the steelhead and salmon take analyses.

Juvenile yelloweye rockfish are likely present within the action area. They do not typically use the nearshore but occupy the shallow range of adult habitats at greater than 120 feet in depth (Love et al. 2002). Juvenile yelloweye rockfish are not expected to occur in the mixing zone (very infrequently if at all) as it only extends to 108 feet in depth and densities are expected to be very low. Additionally, what is presumed to be a gradually sloping mud bottom is not their preferred habitat.

Juvenile canary rockfish and bocaccio are likely present within the action area given the presence of patchy kelp in some areas. But, as is the case with juvenile yelloweye rockfish, densities are expected to be very low. Juvenile bocaccio and canary rockfish settle onto shallow nearshore waters in habitats that support the growth of kelp at 3 to 6 months of age. They then move progressively to deeper waters as they grow during the late fall and winter months (Love et al. 2002). Indications are there is a general lack of eelgrass, kelp and rocky habitats within the mixing zone and just minimal amounts in the overall action area (Paulson 2009). Specific habitat information is lacking for the mixing zone and surrounding area. If this data were available a much more informed conclusion could be determined regarding potential use of the mixing zone by juvenile canary rockfish and bocaccio. As a result juvenile canary and bocaccio are not expected to use the mixing zone area (very infrequently if at all) as rearing habitat.

Adult yelloweye rockfish, canary rockfish and bocaccio typically occupy waters deeper than 120 feet (Love et al. 2002). These deep-water habitats include extreme slopes of unconsolidated substrates, or sand, shell, and cobble fields often located in the periphery of rocky outcroppings. These deep unconsolidated habitats occur off many of the islands and Points of South Sound such as Tacoma Narrows, Fox and Ketron Islands. While these three species of rockfish are most likely present within the action area there is only a slight chance they could occasionally be found within the mixing zone. Adult yelloweye rockfish are for the most part unlikely to occupy demersal habitat within the mixing zone do to the relative lack of depth, structural complexity and steepness. Adult yelloweye rockfish are generally structure oriented, remain near the bottom and have small home ranges (Love et al. 2002).

A few adult canary rockfish or bocaccio could stray through the water column within or near the mixing zone because some individuals have larger home ranges, move long distances, and spend time suspended in the water column (Love et al. 2002). Adults of these two species are most commonly found in waters deeper than 120 feet, (Love et al. 2002, Orr et al. 2000). Given the low abundance of adult canary rockfish and bocaccio, it is unlikely that they would be traveling though the affected water column (which is less than 0.0004 percent of the Puget Sound) within the mixing zone. Also, the general lack of depth and general habitat conditions (structural complexity and steepness) are not conducive to utilization by these species. There is no specific information for the mixing zone and surrounding areas regarding habitat conditions. If these data were available, a much more informed conclusion could be determined regarding potential use of the mixing zone by adult canary rockfish and bocaccio.

Effluent discharge could affect rockfish by altering their food sources. Some rockfish prey are probably injured or killed by the plume in addition to contaminants accumulating in benthic habitats. Contaminants adhering to suspended sediments could eventually accumulate in the rockfishes' food. Effects to food quality and bioaccumulation could persist beyond the five years of the implementation of the proposed action.

Adult, larval, and rearing juvenile surf smelt, Pacific sand lance, and herring are present in the action area throughout the year, and likely use habitat within the mixing zone from time to time. Other life-stages of potential rockfish food (e.g., flatfish and other species) also use habitat in the affected water column or demersal habitats at the outfall mixing zone site (Dinnel et al. 1986; Donnelly et al. 1988). As the larvae and juveniles from these forage fish species encounter the mixing zone some would eventually be injured or killed within the water column by methods similar to those for rockfish larvae and salmon. Because forage fish move throughout the Puget

Sound, the loss of some forage fish in one area could eventually reduce prey availability in other areas.

Southern Resident Killer Whales

The proposed project will cause reduced prey quality and quantity, and accumulation of toxic chemicals in the whales. There are no direct effects from the proposed action. This section evaluates the indirect effects of the proposed action on the Southern Resident killer whale DPS as well as the effects of other activities that are interrelated or interdependent with that action, and determines how the effects of the proposed action, and of interrelated and interdependent actions, interact with the environmental baseline (50 CFR 402.02). The analysis below estimates the reduced prey quality and prey availability, and provides the results from the incremental PBDE accumulation associated with the proposed action.

Effects of Reduced Prey Availability and Quality. Chinook salmon are the primary prey of Southern Residents during the spring and summer months while they are feeding in the inland water of Puget Sound and the Georgia Basin (see further discussion in Section 2.2, Status of the Species). This section therefore focuses on the effects of the proposed action on Chinook availability and quality in inland waters of the Puget Sound. We primarily focus on the presence of PBDEs and the exposure of this toxin to the aquatic food web because they are found in measurable and significant amounts in wastewater effluent, have no protective regulations, and may interact synergistically with contaminants currently in high concentration in the whales (e.g., PCBs) and pose a risk to the Southern Resident killer whales. NMFS anticipates that Southern Residents will be exposed to toxic chemicals, such as PBDEs, through ingestion of prey that are either directly exposed to the Solo Point wastewater effluent or from the exposure of the pelagic food web at lower trophic-levels, which would biomagnify up the food chain and could potentially result in a higher risk to the whales.

Reduced Prey Availability. Over the long term, the action is likely to continuously deliver PBDEs to the aquatic environment through discharged effluent (a point source), which eventually leads to the contamination of the aquatic food web. This raises concern because baseline levels for PBDEs in outmigrant juvenile Chinook salmon sampled in the Puget Sound are comparable to those associated with biological effects such as immuno-suppression, oxidative stress, neurodevelopmental toxicity, or thyroid hormone alterations (Arkoosh et al. 2010; Sloan et al. 2010). Currently there is no health effects threshold or tissue residue guideline identified for PBDEs in adult Chinook salmon. It is currently unknown if baseline levels of PBDEs in adult Chinook salmon are high enough to reduce prey availability and food supply for the Southern Resident killer whales.

Mortality of Chinook could reduce prey availability to the whales where the marine ranges of the affected Chinook stocks and the whales overlap. As described in Section 2.4.1 above (see Table 13), the proposed action is likely to cause mortality of juvenile salmonids, which could affect the Southern Resident's prey availability in future years. The estimated annual reduction of 7,371 juveniles from exposure to the mixing zone or increased predation would translate to an effective loss of no more than 38 adult Chinook salmon from a variety of runs (see Table 13) across a 3-5 year period (i.e., by the time these juveniles would have grown to be adults and available prey of killer whales).

Given the total quantity of prey available to Southern Resident killer whales throughout their range, this annual reduction in prey is extremely small, and although measurable the percent reduction in prey abundance is not anticipated to be different than zero by multiple decimal places (based on NMFS previous analyses of the effects of salmon harvest on Southern Residents; e.g., NMFS 2008e; NMFS 2011b). Because the reduction is so small, there is also a low probability that any of the juvenile Chinook salmon killed by the proposed action would have later (in 3-5 years time) been intercepted by the killer whales across their vast range in the absence of the proposed action. Therefore, the anticipated annual reduction of Chinook associated with the proposed action would result in an insignificant reduction in adult equivalent prey resources for Southern Resident killer whales.

Reduced Prey Quality. As discussed in the Status of the Species, the quality of Chinook salmon is likely influenced by a variety of factors, including contaminant load. The proposed project may affect listed Southern Resident killer whales by indirect effects from reduced prey quality through the accumulation of toxic chemicals, particularly PBDEs. The accumulation of PBDEs in prey will result in bioaccumulation and biomagnification in the whales. Below we compare the results of the first scenario (which includes federal and non-federal sources for PBDEs with the exception of Solo Point WWTP) with the results from the second scenario (which includes all sources with the Solo Point WWTP) as described in the Environmental Baseline.

Incremental Increase Model: PBDEs in the Effluent. The primary steps 1 through 4 for the incremental increase model are described in the Environmental Baseline. In step 4 of the first scenario, we evaluated PBDE concentrations in two individual whales from J pod (an adult male and a male calf) over a biologically meaningful time frame without a contribution of PBDEs from the discharged effluent from the Solo Point WWTP (see Figure 14 for results from the first scenario). The adult male increased its PBDE concentration from an initial measured value of 6,300 ng/g lipid blubber weight (Krahn et al. 2007b) to over 15,000 ng/g over the biologically meaningful time frame we evaluated. The calf had had an initial measured PBDE value of 15,000 ng/g (Krahn et al. 2007) and was predicted to experience a growth dilution followed by an increase in contamination.

In the remainder of this section, we discuss the second scenario in which we include PBDE loadings from the Solo Point WWTP and compare the two scenarios in order to isolate the effects from the Solo Point WWTP. The development of the model parameters are described in more detail below and include (1) the estimated PBDEs in the Solo Point wastewater effluent, (2) PBDEs in resident Chinook salmon in both scenarios, and (3) the consequential PBDE accumulation in two individual killer whales from J pod (an adult male and a male calf).

Estimated PBDEs in the Solo Point Wastewater Effluent. There is currently no data available for PBDE levels in the Solo Point WWTP effluent. Therefore, we identified the Bremerton sewage treatment plant as a surrogate WWTP with known PBDE levels that had similar mgd and treatment processes to those of the Solo Point facility. We used this surrogate to estimate the fraction of PBDE input into Puget Sound from Solo Point. The Bremerton sewage treatment plant receives domestic sewage from residential and light commercial activities in the city of Bremerton. The plant also receives wastewater from a hospital and domestic and industrial pretreated wastewater from Puget Sound Naval Shipyard (PSNS). The plant consists of an activated sludge secondary treatment system with an average maximum flow of 10.1 mgd and a representative flow of 4.30 mgd. Total PBDEs in effluent from this facility were measured in February and July of 2009 and the maximum PBDE concentration was 17,069 pg/L (WDOE and Herrera Environmental Consultants, Inc 2010). We converted this maximum concentration measurement (in pg/L) to a maximum annual total PBDE concentration in the effluent (in kg/yr) using the average design flow of 7 mgd. We assumed this concentration was the same in the Solo Point effluent and equaled 0.165 kg/yr.

Concentration of PBDEs in the Resident Chinook Salmon in the First and Second Scenarios. We assumed that migratory Chinook salmon have the same PBDE concentrations under both scenarios (6.22 ng/g, see Table 4, Chinook salmon average PBDE concentration), because the majority of their burden is accumulated in coastal waters. In the second scenario, the PBDE concentration in resident Chinook salmon is the current measured value of 40 ng/g (O'Neill et al. 2006). In the first scenario described in the Environmental Baseline section, we assumed that there would be no contribution from Solo Point and therefore, PBDE loadings to Puget Sound would be slightly lower than the current measured values. Assuming that PBDE concentrations in resident Chinook salmon were reduced by the same proportion as total Puget Sound loadings, the PBDE burden in the average resident Chinook was reduced by 0.45 percent from 40 ng/g to 39.82 ng/g.

Estimated PBDEs in Two Southern Resident Individuals. Figure 16 shows the difference in PBDE concentrations for two individuals between the contributions from the Solo Point WWTP as compared to loadings estimated in the first scenario during the 5-year permit. The differences in PBDE concentrations increased linearly in the time frame we evaluated in both individuals (Figure 16). The cumulative difference in the total PBDE concentration specific to Solo Point from the 5 years worth of discharge covered by this NPDES permit was estimated to be 19 ng/g lipid weight for the adult male and 15 ng/g lipid weight for the male calf. This is equivalent to 18,103 µg of PBDEs in the adult male and approximately 9,089 µg of PBDEs in the calf in the 5-year period, adding to the long-term accumulation that the whales will experience. Each additional permit following the current permit (which is reasonably likely to occur) will continue to add PBDEs into the Puget Sound and increase the cumulative difference in the total PBDE concentration specific to Solo Point.

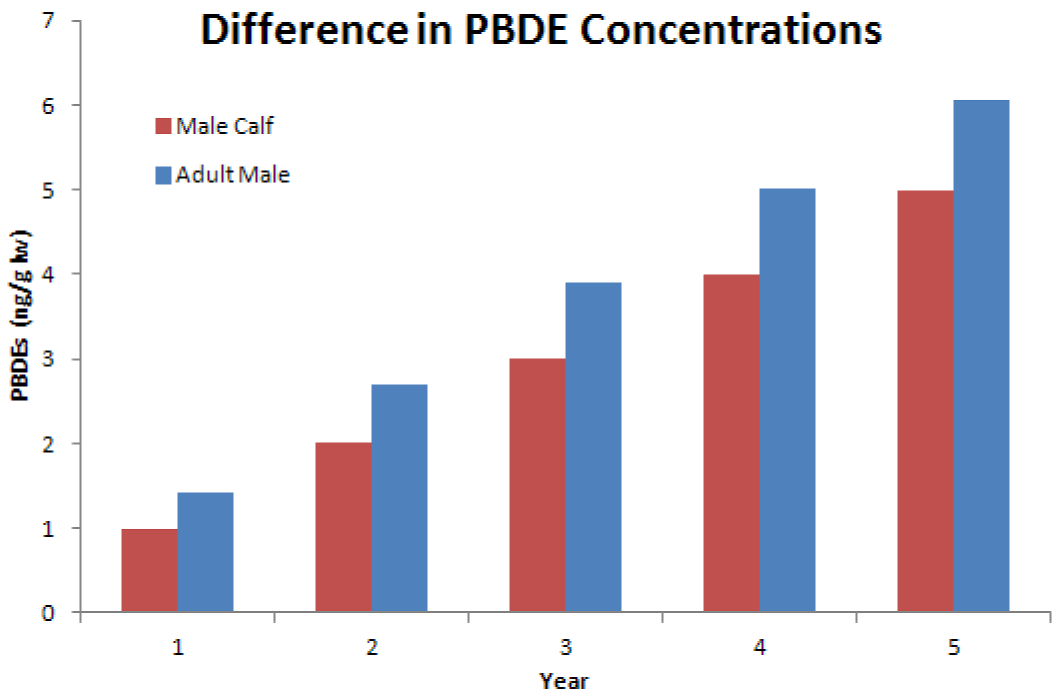

Figure 16. Differences in PBDE concentrations between scenarios in the adult male and male calf. These differences represent the whales' increase in PBDE concentration contributed by the Solo Point WWTP.

The Southern Resident killer whales have PCB levels that are well above a PCB health effects threshold identified for harbor seals (17,000 ng/g described previously). Additionally, PBDE levels in the Southern Residents are already above levels associated with endocrine disruption in juvenile grey seals (Hall et al. 2003). Based on the methods described above, we anticipate that the Southern Resident killer whales would incur adverse health effects over a shorter period of time than would otherwise occur absent the action. Furthermore, there may be synergistic effects between PBDE and PCB congeners likely increasing the health risk to the whales. Thus, increasing PBDE levels in the whales only further exacerbates their current susceptibility to adverse health effects including effects to the whales' reproductive, endocrine, and immune systems.

2.4.2 Effects on Critical Habitat

The analysis does not use the regulatory definition of "destruction or adverse modification" at 50 CFR 402.02 in this Opinion. Instead, it relies on statutory provisions of the ESA, including those in section 3 that define "critical habitat" and "conservation," in section 4 that describe the designation process, and in section 7 that set forth the substantive protections and procedural aspects of consultations, and on agency guidance for application of the "destruction or adverse modification" standard (Hogarth 2005).

Puget Sound Chinook Salmon Critical Habitat

Designated critical habitat for Chinook salmon within the action includes all shoreline areas out to a depth of 30 meters depth with its associated essential physical and biological features. Critical habitat has not been designated for Puget Sound steelhead or the Georgia Basin/Puget Sound rockfish species listed under the ESA. Critical habitat features include, but are not limited to, space for individual and population growth and for normal behavior; food, water, air, light, minerals, or other nutritional or physiological requirements; cover or shelter; sites for breeding, reproduction, and rearing of offspring; and habitats that are protected from disturbance or are representative of the historical geographical and ecological distribution of a species (NMFS 2005).

The principal biological or physical elements that are essential to the conservation of the species in the Puget Sound ESU are known as primary constituent elements (PCEs). Primary constituent elements include, but are not limited to: spawning sites, feeding sites, seasonal wetland or dry land, water quality or quantity, host species or plant pollinator, geological formation, vegetation type, tide, and specific soil types to be in good functioning condition, PCEs of nearshore marine areas are to be free of obstruction and excessive predation with: "(i) water quality and quantity conditions and forage, including aquatic invertebrates and fishes, supporting growth and maturation; and (ii) natural cover such as submerged and overhanging large wood, aquatic vegetation, large rocks and boulders, and side channels. Nearshore marine features are essential to conservation because without them juveniles cannot successfully transition from natal streams to offshore marine areas (NMFS 2005).

As part of its assessment the CHART considered the conservation value of each watershed in the context of the populations within the five geographic regions of diversity and correlated risk in Puget Sound identified by the Puget Sound TRT (2002). The team evaluated nearshore marine areas for this ESU focused on this area because it generally encompasses photic zone habitats supporting plant cover (e.g., eelgrass and kelp) important for rearing, migrating, and maturing Chinook salmon and their prey. The CHART concluded that habitat areas in all 19 nearshore zones of Puget Sound (including areas adjacent to islands), Hood Canal, and the Strait of Juan de Fuca (to the mouth of the Elwha River) warrant a high rating for conservation value to the ESU. These habitat areas are found along approximately 2,376 miles of shoreline within the range of this ESU (NMFS 2005).

The essential elements of the nearshore marine PCE affected by the proposed action are water quality (free of obstruction and excessive predation) and forage (including aquatic invertebrates and fishes), supporting growth and maturation. The water quality degradation caused by this discharge creates a barrier to migration and forces juvenile fish to deeper water adversely affecting the conservation role of the nearshore marine PCE in the action area. Forcing the fish into deeper water undermines the freedom from predation element of the nearshore PCE in the action area as juvenile salmonids are more susceptible to predation in deeper water. Furthermore, migrating salmonids will avoid the mixing zones within seven acres of the action area; impairing the role the area plays in supporting growth and maturation of the Chinook salmon for which the habitat was designated critical. Finally, these areas and those surrounding the mixing zones in the action area provide decreased forage quality and quantity as the result of altered plant and animal assemblages in these areas (Johnson et al. 2009).

Puget Sound net flows and circulation patterns create conditions where persistent toxic contaminants remain in certain areas of the Sound for long periods of time. The association of toxic contaminants with effluent suspended solids, combined with extended residence times in sound waters, allows contaminants to settle into sediments where organisms are exposed. Animals that rely on organisms associated with sediments are at risk of toxic contamination through this process. In this case, sediments and benthic invertebrates are contaminated which affects forage fish, which in turn are also a food item for juvenile Chinook salmon. With documented surf smelt spawning in the close vicinity, impaired water quality would affect use of that habitat, eggs that may be deposited, larval and adult life history stages. Sand lance and herring are also important food sources for Chinook salmon and would also suffer effects from impaired water quality in and surrounding the mixing zones.

Southern Resident Killer Whale Critical Habitat

The proposed action occurs within and affects critical habitat designated for Southern Resident killer whales. Based on the natural history of the Southern Residents and their habitat needs, we identified three physical or biological features essential to conservation in designating critical habitat: (1) Water quality to support growth of the whale population and development of individual whales, (2) Prey species of sufficient quantity, quality and availability to support individual growth, reproduction and development, as well as overall population growth, and (3) Passage conditions to allow for migration, resting, and foraging. This analysis considers effects to these features.

Effects of permitting the Solo Point WWTP include reduced quantity and quality of prey that can support individual growth, reproduction and development. For reasons described above, it is likely that the anticipated reduction in prey quantity caused by the action would have an insignificant effect on Southern Resident killer whales. Effects on water quality over time with the continued discharge of wastewater effluent which contains persistent bioaccumulative pollutants, including PBDEs will result in reductions in prey quality in the action area. The biomagnification of PBDEs in the aquatic food web, including Chinook salmon, and the bioaccumulation in individual whales is likely to persist and increase. As described previously, the proposed action is likely to result in PBDE accumulation at a faster rate than without the proposed action in Southern Residents' primary prey, (particularly in adult resident Chinook salmon). This level of PBDE accumulation over the 5-year permit time frame is likely to increase the body burdens in individual whales by small but measurable amounts.

We do not expect the proposed NPDEs permit to affect passage because neither the effluent nor the outfall from the wastewater treatment plant is a barrier to passage.

2.5 Cumulative Effects

"Cumulative effects" are those effects of future state or private activities, not involving Federal activities, that are reasonably certain to occur within the action area of the Federal action subject to consultation (50 CFR 402.02). Future Federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation pursuant to section 7 of the Act.

Overall, Central and South Puget Sound have a very high percentage of shoreline armoring, and a large number of overwater structures. The major urban areas of South Puget Sound (Burns 1985) are located in the western portions of Pierce County, Olympia in Thurston County, and Shelton in Mason County. Of these four counties Thurston County exhibited the strongest growth most recently. Among all counties in Washington State it has experienced the third highest growth in the last ten years, over 20 percent (Table 17).

Table 17: Population Growth[5]

County	Population in 2000	Population in 2010	Population Change	Percent	Rank
Pierce	700,818	814,600	113782	16.235599	11
Thurston	207,355	252,400	45045	21.7236141	3
Mason	49,405	57,100	7695	15.5753466	13
King	1,737,046	1,933,400	196354	11.3039033	19

Future private and state actions are reasonably certain to continue within the action area, increasing as population density rises. Anticipated negative effects from future development include continuing contribution to overall reduced water quality (temperature, nutrients, contaminants, among other elements of water quality), continuing modification of the existing altered hydrograph associated with increasing impervious surface from build out, additional simplification of shoreline characteristics, all of which are likely to degrade salmon habitat in the South Puget Sound region. Human-induced stressors identified by the South Puget Sound Salmon Recovery Group (Shared Strategy 2007) as having the most significant impact on natural processes, and greatest prevalence throughout South Puget Sound, include overwater structures, shoreline armoring, placement of fill below mean high higher water, riparian vegetation loss, wetland and estuarine modification, predation, and boat traffic. All of these are reasonably expected to increase over time with human population growth increases, despite regulatory regimes intended to slow or limit their adverse impacts.

Water quality in the action area will be diminished over time with continued discharges of stormwater runoff and contaminants by public, private and industrial outfalls. The Washington State Department of Ecology provided an initial list of 199 state-regulated facilities or outfalls that discharged wastewater, as well as 54 U.S. EPA-regulated facilities, for a total of 253 permitted facilities for use in developing a database of point source discharges to Puget Sound (WDOE 2008b). The list included both direct discharges to Puget Sound as well as those facilities that discharged to lakes, streams, or rivers that in turn discharged to Puget Sound. Bioaccumulation of contaminants in the aquatic food web is likely to persist, with associated impacts on the marine environment and dependent aquatic organisms.

[5] Data from Office of Financial Management, http://www.ofm.wa.gov/pop/april1/default.asp

Redman et al., (2006) state that future risks include present-day loadings of toxic contaminants from a variety of sources combined with historic loads to create the patterns of contamination and continued impacts that we observe today. Loadings from some sources may grow in future years as a result of increases in population of the Puget Sound basin; the numbers of vehicles in the basin; and the amount of fossil fuel combustion in the basin to provide electricity, heat, and to power motor vehicles. For example, as the population in the Puget Sound region increases the quantity of pharmaceuticals released to sewage systems will also increase. Projected future loadings are difficult to estimate, but percentage increases commensurate with population growth might provide our best projections in the absence of difficult-to predict technological advances (e.g., cleaner burning engines, less harmful pesticides) and societal shifts (e.g., improved methods of disposing of unused pharmaceuticals, decreased vehicle miles traveled as traffic congestion worsens). Although the Puget Sound Partnership may make progress toward reducing marine pollution (Puget Sound Partnership 2008), measurable change is not reasonably certain to occur in the near term. It is reasonably certain, however, that the Partnership's current action plan will be further refined over the next five years (http://www.psp.wa.gov/action_agenda_2011_update_home.php).

According to a recent US Government Accountability Office report, over 700 new chemicals are introduced into commerce every year (e.g., as flame retardants, pesticides, additives in the manufacture of plastics, pharmaceuticals) and current practices do not adequately assess chemicals' risks before they enter commerce. The use and intended or unintended release of these chemicals to the environment may pose additional toxic risks to the Puget Sound ecosystem (Redman et al. 2006).

As the human population in the action area continues to grow, demand for agricultural, commercial, and residential development will grow as well. Land use changes and development of the built environment are likely to continue under existing zoning. Population growth in South Puget Sound will continue to exert pressure on remaining viable habitats for listed species. The NMFS believes that many of the existing local and state regulatory mechanisms intended to minimize and avoid effects on subbasin function and listed species from future development are generally not adequate or not implemented sufficiently. Though these existing regulations could decrease adverse effects, as currently constructed and implemented, they still allow incremental degradation to occur. Over time, that incremental degradation, when added to the already degraded environmental baseline, can result in reduced habitat quality for at-risk salmon, steelhead rockfish, and Southern Resident killer whales.

Many of the growth-related negative effects mentioned above will have no federal nexus and not be consulted on individually. However, those actions will greatly reduce the conservation value of critical habitat for Puget Sound Chinook salmon and Southern Resident killer whales for as long as the human population increases.

In addition to growth, sea level rise will be exacerbated in South (the action area) and Central Puget Sound by downward vertical land movements of up to 2 millimeters per year (WDOE 2006). This will likely mean increased pressure to conduct shoreline fill and armoring.

Final Recovery Plan for Southern Resident Killer Whales was published January 24, 2008 (NMFS 2008a), and the Puget Sound Salmon Recovery Plan was adopted on January 19, 2007.It was developed through the Shared Strategy for Puget Sound, a collaborative conservation effort that includes state, tribal and local governments, industry, conservation groups and others. The plan lays out long-term recovery goals and strategies, but its primary focus is on the next ten years of actions to place this region on a path toward recovery. This is because its ultimate success depends upon the various authorities and responsible parties stepping up to commit to implement the strategies and actions described in the plan.

Although state, tribal and local governments have developed plans and initiatives to benefit marine fish species, ESA listed salmon, and the listed Southern Resident killer whales, NMFS cannot consider them reasonably certain to occur in its analysis of cumulative effects until more concrete steps are taken in their implementation. Government actions are subject to political, legislative and fiscal uncertainties. These realities, added to geographic scope of the action area, which encompasses several government entities exercising various authorities, and the changing economies of the region, makes analysis of cumulative effects difficult. There are some impacts that we predict are reasonably certain to occur into the future, such as construction and other habitat altering activities, and marine pollution, as discussed above.

Other beneficial actions include WDFW adopting a series of measures to reduce death of rockfish from non-tribal fisheries within the Puget Sound/Georgia Basin. These measures include: A closure of the entire Puget Sound to retention of any rockfish species; Prohibition of fishing for bottomfish deeper than 120 feet; Closure of the set net and set line fishery; Closure of the bottom trawl fishery; Closure of the inactive smelt purse seine fishery; Closure of the inactive scallop trawl fishery; Closure of the inactive pelagic trawl fishery; and Closure of the inactive bottomfish pot fishery.

These measures will eliminate direct harvest of rockfish, and reduce or prevent by-catch from non-tribal recreational and commercial fisheries within the U.S. portion of the Puget Sound/Georgia Basin that were factors of decline for yelloweye rockfish, canary rockfish and bocaccio (Drake et al. 2010).

Cumulative effects are likely to have a long-term, adverse effect on Puget Sound Chinook salmon, Puget Sound steelhead, rockfish, and Southern Resident killer whale population abundance and productivity. To the extent that recovery actions are implemented and on-going actions regulatory mechanisms continued, adverse cumulative effects may be minimized, but will not be completely avoided.

2.6 Integration and Synthesis

The Integration and Synthesis section is the final step of NMFS' assessment of the risk posed to species and critical habitat as a result of implementing the proposed action. In this section, we add the effects of the action (Section 2.4) to the environmental baseline (Section 2.3) and the cumulative effects (Section 2.5) to formulate the agency's biological opinion as to whether the proposed action is likely to: (1) result in appreciable reductions in the likelihood of both survival and recovery of the species in the wild by reducing its numbers, reproduction, or distribution; or

(2) reduce the value of designated or proposed critical habitat for the conservation of the species. These assessments are made in full consideration of the status of the species and critical habitat (Section 2.2).

Puget Sound Chinook Salmon ESU

For the next five years, this action permits the continuation of existing, ongoing discharge of a plume of contaminants at concentrations that create an avoidance response in most of the fish that encounter it. The presence of a plume evoking avoidance by exposed fish creates two results: 1) the presence of a migratory barrier that diverts juvenile Chinook salmon out into deeper water where, 2) predation rates can be increased by as much as fivefold (Willette 2001). The consequence of avoiding the plume of contaminants is continued death by predation of some juvenile salmonids, for the duration of the permit. Some fish will not avoid the plume. Therefore, direct exposure to contaminants in the effluent plume is likewise an existing source of fish death from impaired olfaction as a more acute effect of exposure, and illness and/or organ damage as a later-developing chronic response. Indirect effects of the proposed action related to bioaccumulation and degradation of prey resources will also likely result in additional delayed death through illness or organ damage. Some subset of salmonids exposed to effluent, either directly or indirectly, is already and will continue to experience merely injury in the form of illness, reduced fitness, or reduced fecundity, rather than death.

Of the 12 Chinook salmon populations affected by this action, eight are ranked as Tier 1 in the draft Puget Sound Chinook Salmon Population Recovery Approach. Populations with that ranking are considered to be the most important for preservation, restoration, and ESU recovery. NMFS has derived rebuilding escapement thresholds for some of the Puget Sound Chinook salmon populations based on an assessment of current habitat and environmental conditions. The rebuilding threshold is defined as the escapement that will achieve Maximum Sustained Yield under current environmental and habitat conditions (NMFS 2000b). Rebuilding thresholds represent a level of spawning escapement, consistent with current environmental conditions, that is associated with rebuilding populations to recovery.

All six populations in the Skagit River Basin are Tier 1. Less than one percent of the juvenile Chinook salmon from those populations is expected to use the action area (Table 12). These populations are managed primarily for natural-origin production. The geometric mean escapement of natural origin spawners was above the rebuilding escapement threshold for the Upper Skagit, Lower Skagit, Upper Sauk River and Lower Sauk River populations (Table 18). The geometric mean escapement of natural origin spawners was below the rebuilding threshold (but well above the critical threshold) for the Suiattle River and the Upper Cascade River populations (Table 18). The number of fish from the Skagit River Basin injured or killed from the NPDES authorized discharge is two adult equivalents out of 14,539 adults (Table 12) or 0.015 percent per year (Table 13). Assuming these numbers remained constant over the 5 year period, this would yield an aggregate reduction in the adult equivalent population of 0.075 percent. The median growth rate for natural origin escapement in all of the Skagit River populations is at 1.0 or above with the exception of the Suiattle River at 0.99 (NMFS 2010a).

The Skykomish and Snoqualmie River populations (Snohomish River Basin) are Tier 2 and 3 respectively. Less than 11 percent of juvenile Chinook salmon from these populations use the action area (Table 12). The geometric mean escapement of natural origin spawners was below the escapement rebuilding threshold (but well above the critical threshold) for the Skykomish River population and above the escapement rebuilding threshold for the Snoqualmie River population (Table 18). The number of fish from the Snohomish River Basin (Skykomish and Snoqualmie) expected to be injured or killed as a consequence of the proposed action is four adult equivalents out of 4,309 or 0.1 percent per year (Table 13), and a 0.5 percent reduction over the full 5 years. The median growth rates for natural origin escapement in these two populations (Skykomish River 1.05 and Snoqualmie River 1.03) are both over 1.0 (NMFS 2010a).

The Green/Duwamish River population is considered a Tier 2 population in the recovery plan. Based on index scores alone, the Central/South Sound region would have only Tier 1 and Tier 3 populations. Therefore, to ensure that at least one population in the region is recovered at a sufficient pace to allow for its potential inclusion as a “Tier 1” population if needed for recovery, the “Tier 3” population with the highest total index score in the Central/South Puget Sound biogeographical region (i.e., Green/Duwamish) was assigned a “Tier 2”ranking (NMFS 2010b). NMFS anticipates that less than 15 percent of juvenile Chinook salmon from this population uses the action area (Table 12). The geometric mean escapement of natural origin spawners was below the escapement rebuilding threshold (but well above the critical threshold) for this population (Table 18). The number of fish from this population expected to be injured or killed by the proposed action is five adult equivalents out of 3,615 or 0.13 percent per year (Table 13). This would extrapolate to a 0.65 percent reduction in the adult population over the 5 year life of the permit. The median growth rate for natural origin escapement in this population is 1.04 (NMFS 2010a).

The White River population is a Tier 1 population. This population is the only early returning (spring-run) adult fish in Central/South Sound. These fish have been identified as one of the two populations in that region which must achieve viability in the ESU to recover the species. NMFS estimates that less than 24 percent of juvenile Chinook salmon from this population uses the action area (Table12). The geometric mean escapement of natural origin spawners was slightly under the escapement rebuilding threshold (but well above the critical threshold) for this population (Table 18). The number of fish from this population expected to be injured or killed by the proposed action is two adult equivalents out of 987 or 0.24 percent per year (Table 13). This extrapolates to a 1.2 percent reduction in the total number of adults over the 5 year life of the project. The median growth rate for natural origin escapement in this population is 1.12 (NMFS 2010a).

The Puyallup River population is Tier 3 and less than 24 percent of juvenile Chinook salmon from this population likely use the action area (Table 12). The geometric mean escapement of natural origin spawners is a little less than the escapement rebuilding threshold (but well above the critical threshold) for this population (Table 18). The number of fish from this population expected to be injured or killed by the proposed action is two adults out of 969 or 0.24 percent per year (Table 13). This would create a 1.2 percent loss in the total number of adults over the 5 year life of the permit. The median growth rate for natural origin escapement in this population is 0.91 (NMFS 2010a).

The Nisqually River population is Tier 1 and NMFS estimates that 45 percent of juvenile Chinook salmon from this population uses the action area (Table 12). The geometric mean escapement of natural spawners (includes naturally spawning hatchery fish) exceeded the escapement rebuilding threshold for this population (Table 18). The number of fish from this population expected to be injured or killed by the proposed action is 22 adults out of 1,549 or 1.2 percent per year (Table 13). These numbers being assumed constant over the 5 year period, would result in a 6 percent total reduction in the adult population. The median growth rate for natural origin escapement in this population is 1.01 (NMFS 2010a).

Table 18. Escapement of Natural Origin PS Chinook Spawners[1]

Population	Geometric Mean Escapement	Critical Escapement Threshold	Rebuilding Escapement Threshold
Upper Skagit (1999-2009)	10,561	967	7,464
Lower Skagit (1999-2007)	2,248	251	2,182
Upper Sauk (1999-2007)	425	130	330
Lower Sauk (1999-2008)	690	200	681
Suiattle (1999-2008)	317	170	400
Cascade (1999-2008)	298	170	1,250
Skykomish (1999-2008)	2,578	1,650	3,500
Snoqualmie (1999-2008)	1,731	400	1,250
Green/Duwamish (1999-2009)	3,615	835	5,523
White (1999-2009)	987	200	1,100
Puyallup (1999-2009)	969	200	1,200
Nisqually (1999-2009)	1,549[2]	200	1,200

[1]NMFS (2010a)
[2]Includes naturally spawning hatchery fish

Yearly totals for associated direct and indirect adverse effects (exposure within and beyond the mixing zone, predation) on natural origin Puget Sound Chinook salmon for the above populations are calculated to be 7,371 juveniles, which translates to 38 adults. This represents 0.14 percent adult equivalents of the total number of natural origin adult Chinook salmon within the ESU (NMFS 2010a). Assuming these numbers remain the same over the next five years it would total 36,855 juveniles and 190 adults.

The JBLM WWTP outfall has been in operation for several decades. So, it is likely these adverse effects have been occurring for that same period of time. Even under those circumstances the growth rates for escapement of natural origin spawners of the affected populations are all over 1.0 with the exception of the Suiattle River (0.99) and Puyallup River (0.91). The Suiattle River is a Tier 1 population and is very close to having a median growth rate for escapement of 1.0. The Puyallup River is the furthest from achieving a median growth rate of 1.0 but is a Tier 3 population.

When considering all spawning fish (includes natural origin spawners and hatchery-origin fish spawning naturally) escapement trends on all affected populations are either stable or increasing for the period of 1999-2009 (NMFS 2010a). In particular the White River population shows a significant increasing trend in growth rates for both return and escapement (NMFS 2010a). Additionally, thresholds for escapement that have been identified for rebuilding these populations (under current habitat and environmental conditions) are exceeded in every case with exception of the Lower Sauk River, the Suiattle River, and the White River (Table 18). Taking this status into account, NMFS concludes that the effects of the proposed action are insufficient in magnitude to significantly alter trends.

The effects of the proposed action are added to the habitat effects in the environmental baseline, and to cumulative effects. Currently, the environmental baseline is mostly informed by existing shoreline armoring and ongoing impacts that carry into the water from the upland human population, including the Solo Point WWTP. Restoration of the large Nisqually estuary has occurred at the edge of the action area and only two functioning pocket estuaries have been identified in the action area. Therefore, estuarine habitat within the action area has been of moderate function because the baseline conditions include extensive shoreline armoring and lack of riparian vegetation. Thus, the existing carrying capacity for rearing smolts in the action area is somewhat diminished. There are cumulative effects associated with shoreline armoring, overwater structures, water quality and climate change. Considering that population growth with continue, the above cumulative effects will persist into the future. The effect of the discharge is more meaningful to those populations that make the most use the action area. For affected fish, the presence of the discharge for the five-year duration of the permit will continue to moderate the function of the action area to support salmonid life histories that are expressed there as it has in the past with no improvement and only slight degradation (to the extent that contaminants bioaccumulate). While we anticipate that the proposed action will continue to adversely affect small number of fish in the affect populations, the level of this effect will not influence the status of these populations. Thus, the effects of the action taken with the effects in the baseline and cumulative effects will not influence existing population viability. Because the effects of the proposed action are not expected to significantly affect population trends among the PS Chinook stocks exposed to the action that contribute to the viability of the PS Chinook ESU, the overall effects of the action will not jeopardize the existence of the PS Chinook ESU or appreciably reduce the likelihood of survival and recovery of this ESU.

Puget Sound Steelhead DPS

For Puget Sound steelhead the magnitude of effects is even smaller than for Puget Sound Chinook, because the numbers making use of the nearshore in the action area are much smaller. Also, based on life history and out-migration patterns, Puget Sound steelhead will not be affected by increased predation on juveniles that may avoid the mixing zone. The juveniles, as they enter the estuary, are generally larger in size and don't utilize the nearshore to the same extent as Chinook, meaning the rate of exposure will be smaller, the avoidance pattern will be reduced, and the risk of predation in deeper water is lower.

For the most recent five years (2005-2009) the geometric mean estimate of adult escapement for Nisqually River origin natural spawners was 402 fish (Ford et al. 2010). Yearly totals for associated indirect and direct mortalities (exposure within and beyond the mixing zone) on natural origin Puget Sound steelhead for the Nisqually River population is calculated to be 60 juveniles and 0.215 adult equivalents (Table 15). This number of adults represents 0.05 percent of this population and 0.001 percent adult equivalents of the entire DPS (WDFW 2008). Assuming these numbers remain the same over the next five years it would total 300 juveniles and 1 adult steelhead. Thus, NMFS concludes that the adverse effect of the discharge, taken together with the baseline and cumulative effects, is not expected to appreciably reduce the likelihood of survival or recovery of the Puget Sound DPS. The production rate of the Nisqually River steelhead population anticipated to be exposed to the effects of the action is adequate for sustained persistence of the population and the proposed action does not significantly change the production rate of this population such that the overall recovery of the Puget Sound DPS will similarly be unaffected.

Puget Sound Chinook Critical Habitat

The entire mixing zone is located in critical habitat for Puget Sound Chinook salmon. Under the environmental baseline, the effluent discharged from the Solo Pt WWTP and other WWTPs affects water quality, availability of forage, and impairs free passage, each of which are essential elements of either the estuarine and marine nearshore PCEs present in the action area. Additionally, nearshore habitat for rearing and migrating fish has been impacted by shoreline armoring and overwater structures. Under the proposed action, effects from the Solo Pt. discharge will continue for five years. Considering future population growth and climate change, there will continue to be private and state actions that will produce cumulative effects associated with those identified above. Taken with the effects in the environmental baseline and cumulative effects, the effects of the action will not change the conservation value of critical habitat in Puget Sound because the habitat in the mixing zone will not be materially altered by the reissuance of the discharge permit.

Yelloweye Rockfish, Canary Rockfish and Bocaccio

The abundance of ESA-listed rockfish has declined dramatically due to past fishery removals (Drake et al. 2010). Yelloweye rockfish, canary rockfish and bocaccio of the Puget Sound/Georgia Basin are likely affected by existing contaminants and nutrient inputs (West et al. 2001; Palsson et al. 2009). Rockfish productivity is naturally low in most years, and their long life span is an adaptation to this low productivity by enabling multiple years of reproduction (Love et al., 2002). Further evidence of their naturally low productivity comes from the birth of up to several million larvae per mature female per season, and the extremely low survival rates of larvae in most years (Love et al. 2002; Weis 2004). Their naturally low productivity is likely further compromised by past fishery removals (that reduced larger and older fish) and bioaccumulative chemicals (Drake et al. 2010). These factors lead to concerns about the viability of each species (Drake et al. 2010).

The proposed action is unlikely to harm juvenile and adult ESA-listed rockfish. Due to a presumed lack of proper habitat conditions (eelgrass, kelp, rocky areas, depth, steepness, and structural complexity) they are unlikely to occupy benthic/nearshore habitats in the mixing zone. Most of these habitats are thought to be missing from a good portion of the action area and generally not seen in close proximity of the mixing zone. Fish densities are expected to be very small.

Yelloweye rockfish produce between 1,200,000 and 2,700,000 larvae per year (Love et al. 2002). Canary rockfish produce between 260,000 and 1,900,000 larvae per year (Love et al. 2002). Bocaccio produce between 20,000 and 2,298,000 eggs per year (Love et al. 2002). The estimated number of larvae exposed to the mixing zone on an annual basis would be a maximum of 130,478 yelloweye rockfish, 195,718 canary rockfish, and 4,241 bocaccio. Assuming the effects on rockfish larvae would be similar to those for juvenile Chinook salmon, yearly total take would be equivalent to; 9,786 or 0.36 to 0.82 percent of what one adult yelloweye rockfish produces annually, 14,679 or 0.77 to 5.8 percent of what one adult canary rockfish produces annually; and 318 or 0.01 to 1.6 percent of what one adult bocaccio produces annually. These very low levels of mortality are unlikely to result in an appreciable reduction of the number of larvae that eventually reach adulthood, and therefore are unlikely to further exacerbate the low productivity and abundance of yelloweye rockfish, canary rockfish and bocaccio of the Puget Sound/Georgia Basin. In addition, rockfish larvae naturally experience extremely low survival rates in most years (Love et al. 2002). Juvenile survival rates are generally poor because larval survival and settlement are dependent upon the vagaries of climate, the abundance of predators, oceanic currents and chance events. Long-lived rockfish populations enable persistence through many years of poor reproduction until one good recruitment year (Tolimieri & Levin 2005; Drake et al. 2010).

Our estimate of larvae mortality is probably greater than would take place on an annual basis. The analysis was informed by rockfish larvae data reported in Weis (2004), which occurred only in the San Juan basin. The rockfish population is greater there than in the Puget Sound (Palsson et al. 2009). Thus there are probably fewer ESA-1isted rockfish larvae at Solo Point than we have conservatively estimated. Therefore, fewer larvae are likely to be affected than have been accounted for in the calculations.

The loss of rockfish prey resulting from effluent discharge is unlikely to appreciably alter rockfish feeding opportunities for several reasons. The densities of rockfish in South Puget Sound are very low. The habitat area significantly affected by this outfall is a small percentage of the total amount of benthic habitats in Puget Sound. Therefore the number of rockfish prey killed would constitute only a small fraction of available food sources. Also, rockfish eat many different species of fish and invertebrates (Washington et al. 1978; Lea et al. 1999; Love et al. 2002; Palsson et al. 2009) thus the loss of some of their prey would be unlikely to result in fewer feeding opportunities.

The benthic environment of Puget Sound has been affected by past bottom-trawling, toxicant loading, bioaccumulating contaminants, and the accumulation of derelict fishing gear. Recent and on-going removal of derelict fishing gear has improved benthic habitat suitability for ESA-listed rockfish and their prey base, though nets deeper than 100 feet persist. The cumulative effects of new non-tribal recreational and commercial fishing regulations will further reduce risks to the viability parameters of ESA-listed rockfish.

The proposed action may have a slight effect on rockfish abundance (mortality of a relatively small number of larvae within the mixing zone). It is unlikely to result in juvenile or adult mortality of any individuals that may use the nearshore and/or benthic habitats of the action area. As a result, it is not likely there will be an appreciable reduction in the likelihood of survival or recovery of the species in the wild due to effects on their productivity, diversity, or structure within the Puget Sound/Georgia Basin.

Southern Resident Killer Whales

This section discusses the effects of the action in the context of the status of the species and designated critical habitat, the environmental baseline, and cumulative effects, and offers our opinion as to whether the effects of the proposed action are likely to jeopardize the continued existence of the Southern Residents or adversely modify or destroy Southern Residents' designated critical habitat.

Several factors identified in the final recovery plan for Southern Resident killer whales may be limiting recovery. These are quantity and quality of prey, toxic chemicals that accumulate in top predators, and disturbance from sound and vessels. Oil spills are also a risk factor. It is likely that multiple threats are acting together. For example, nutritional stress can cause whales to draw on fat stores, mobilizing more contaminants that are at relatively high levels and stored in their blubber, affecting reproduction and immune function. Although it is not clear which threat or threats are most significant to the survival and recovery of Southern Residents, all of the threats are important to address. Based on the natural history of the killer whales and their habitat needs, we identified three physical or biological features essential to conservation in designating critical habitat: (1) Water quality to support growth of the whale population and development of individual whales, (2) Prey species of sufficient quantity, quality and availability to support individual growth, reproduction and development, as well as overall population growth, and (3) Passage conditions to allow for migration, resting and foraging.

The Southern Resident killer whale DPS is composed of one small population (88 whales) which is currently at most half of its likely previous size (140 to as many as 400 whales). The effective population size (based on the number of breeders under ideal genetic conditions) of 26 whales is very small, and this in combination with the absence of gene flow from other populations may elevate the risk from inbreeding and other issues associated with genetic deterioration. This population has a variable growth rate (28-yr mean=0.3 percent ± 3.2 percent s.d), and risk of quasi extinction that ranges from 1 percent to as high as 66 percent over a 100-year horizon, depending on the population's survival rate and the probability and magnitude of catastrophic events. Because of this population's small size, it is susceptible to demographic stochasticity and genetic deterioration, as described in the Status of the Species. The influences of demographic stochasticity and potential genetic issues in combination with other sources of random variation combine to amplify the probability of extinction, known as the extinction vortex.

The larger the population size, the greater the buffer against stochastic events. It also follows that the longer the population stays at a small size, the greater its exposure to demographic stochastic risks and genetic risks. In addition, as described in the Status of the Species section, small populations are inherently at risk because of the unequal reproductive success of individuals within the population. The more individuals added to a population in any generation, the more chances of adding a reproductively successful individual. Random chance can also affect the sex ratio and genetic diversity of a small population, leading to lowered reproductive success of the population as a whole. For these reasons, the failure to add even a few individuals to a small population in the near term can have long-term consequences for that population's ability to survive and recover into the future. A delisting criterion for the Southern Resident killer whale DPS is an average growth rate of 2.3 percent for 28 years (NMFS 2008a). In light of the current average growth rate of 0.3 percent, this recovery criterion and the risk of stochastic events and genetic issues described above underscore the importance for the population to grow quickly.

The PBDEs are an emerging concern for killer whales and are potential endocrine disruptors that can affect thyroid hormone levels, can cause subtle neurobehavioral effects, mimic or offset reproductive processes, and alter immune response. Measured PBDE concentrations in the Southern Resident killer whales (Krahn et al. 2007, 2009) are higher than those associated with altered thyroid hormone levels in post-weaned and juvenile grey seals (Hall et al. 2003). Juvenile killer whales have the highest PBDE blubber concentrations (Krahn et al. 2007), which are 10 times higher than concentration levels associated with endocrine disruption in the seals (Hall et al. 2003). Some chemicals can interact at doses below the NOEC level for a single chemical and produce significant effects. Additionally, a non-linear dose-response can occur, such that enhancement of the response occurs at low doses and inhibition occurs at higher doses. Mixture effects can occur at a wide range of doses, and thus, even low concentrations of PBDEs in the whales may combine together with their high concentrations of PCBs and adversely affect the health of the individual Southern Resident killer whales.

The PBDEs are found in significant and measurable amounts in wastewater effluent that discharge into surface waters. The total PBDE loading from wastewater discharge accounts for 25 to 38 percent of the total loading into Puget Sound (including their critical habitat). PBDE levels in the Puget Sound water column are almost 10 times higher than those reported in waters in the Georgia Strait. Once in the water column, PBDEs readily bind to particles and therefore marine sediment can act as a sink and sequester them. Particularly near shore, sediment can also act as a source for aquatic food webs (i.e., a major pathway to killer whales). Sediment PBDE concentrations are typically higher near wastewater outfalls. For example, sediment values ranged between 270 to 1800 pg/g in the Strait of Georgia, however PBDE concentrations in the sediment near a wastewater outfall was 7 to almost 50 times greater than this reported range.

Puget Sound is highly susceptible to toxic input because it is hydrologically isolated, and many species are known to exhibit a high degree of residency and therefore are exposed to more persistent pollutants. For example, resident herring and resident Chinook salmon from the Puget Sound have significantly higher contaminant levels than their northern and coastal counterparts. Southern Residents are also more contaminated than the Northern Resident killer whales, likely as a result from ingesting highly contaminated and localized prey in industrialized areas, such as the inland waters of Puget Sound and Georgia Strait (including their critical habitat).

The action is likely to adversely affect Southern Resident killer whales (including reducing the conservation value of critical habitat) by decreasing prey quality and increasing PBDE levels in the whales. We estimated the incremental increase in the whales' PBDE concentrations caused by the action. The incremental increases over the 5-year permit cycle are small. For example, the total PBDE accumulation specific to Solo Point from the 5 years worth of discharge covered by this NPDES permit was estimated to be 19 ng/g lipid weight for the adult male and 15 ng/g lipid weight for the male calf. This is equivalent to 18,103 μg of PBDEs in the adult male and approximately 9,089 μg of PBDEs in the calf in the 5-year period.

The proposed action reduces the time until PBDE concentrations in individual killer whales will surpass a health-effects threshold (i.e., PBDE accumulation over the lifetime of a killer whale will occur more rapidly with the action than without it). Calves and adult males are at a high risk of adverse health effects from PBDE accumulation, and thus were the focus of our analysis. Although it is not clear if PBDE levels in the Southern Residents are at or near a health-effects threshold, their body burdens are above PBDE concentrations associated with altered thyroid hormones in grey seals. Based on known mixture effects and the similarity in mode of action between PCBs and PBDEs, it is reasonable to assume the whales' are susceptible to synergistic effects between their currently high PCB concentrations and their increasing PBDE concentrations, such that the whales' current levels of both may be sufficient to produce adverse health effects. Thus, increasing PBDE levels in the whales only further exacerbates their current susceptibility to adverse health effects including effects to the whales' reproductive system, the endocrine system, and the immune system. However, the small increase in PBDE levels (19 and 15 ng/g lipid weight in the adult male and male calf, respectively) and potential adverse health impacts are not anticipated to increase the risk of mortality for whales currently in the population during the 5-year permit cycle (and therefore will not rise to the level of serious injury or mortality). It is worth noting that JBLM is expected to experience population growth soon after the 5 year period of this proposed action, and discharge from Solo Point is expected to continue, most likely at a higher rate given this growth. While discharge beyond the term of the current permit is outside of the scope of this proposed action, long-term, continued discharge resulting in increased PBDE accumulation in the whales could reduce population growth.

We have used the best available information to analyze impacts from the Solo Point wastewater effluent on the whales, but acknowledge the limitations of that information. For example, there is currently no data on total PBDE concentrations in the Solo Point effluent. As a conservation measure, JBLM will monitor for PBDEs in the influent and effluent during the permit cycle to fill in this data gap. Additionally, we made several assumptions to develop the incremental increase model, some of which were conservative assumptions that operate to give the benefit of the doubt to the species. For example, assimilation rates and elimination rates of PBDEs in whales are unknown. We assumed 100 percent assimilation and zero elimination because of the lack of data specific to PBDEs in killer whales. This focus of the analysis operates to give the benefit of the doubt to the species (e.g., the estimated PBDE accumulation in the whales might be less than we analyzed if assimilation is actually below 100 percent or if the elimination rate is above zero).

Conservation measures associated with the proposed action may help slow or reverse the trend of PBDE loadings into the Puget Sound. At the scale of the proposed NPDES permit, the EPA and JBLM commit to the following communications that show potential for reducing PBDE levels in Solo Point effluent over the long term. The EPA will request that JBLM consider treatment technologies to minimize PBDE levels in effluent as the base plans to upgrade their facility. Additionally, JBLM will write a letter committing to pursuing funds necessary for plant upgrades, and will share their timeframe for these upgrades, pending funding. Commitment to these communications shows the desire to reduce effects from the discharged effluent.

The EPA put forward additional conservation measures that go beyond the proposed NPDES permit and will address the impacts of PBDEs in wastewater effluent more broadly. These measures are designed to both communicate the current state of knowledge about effects of PBDEs on Southern Residents with other agencies that manage water quality and improve our understanding of these effects. In particular, the inclusion of the Washington Department of Ecology and NMFS, as partners in implementing key elements of a conservation measure to establish a policy forum makes it far more likely that agencies with management responsibilities for water quality will take actions to monitor for and minimize PBDEs in effluent. The EPA shows follow through in this regard by committing to coordinate with NMFS and Ecology on re-issuance of an Ecology-issued NPDES permit to address PBDEs, unless EPA and NMFS determine these issues are being adequately addressed. The EPA will also work with NMFS to establish and convene a technical workgroup that will seek opportunities to fill data gaps and refine uncertainties about the effects of PBDEs on Southern Resident killer whales and wastewater treatment technologies to minimize those effects. These conservation measures will position NMFS to use the new information generated to take actions through subsequent consultation processes. These conservation measures form the basis for a long-term program to seek adjustments to NPDES permits and minimize PBDE loadings if our improved understanding of PBDE effects warrants such adjustments. Such a program is necessary if there is to be a meaningful solution to an issue that is so broad in scope and involves actions outside the jurisdiction of NMFS.

2.7 Conclusion

After reviewing the current status of the listed species, the environmental baseline within the action area, the effects of the proposed action, and cumulative effects, it is NMFS' biological opinion that the proposed action is not likely to jeopardize the continued existence of Puget Sound Chinook salmon or to destroy or adversely modify its designated critical habitat.

After reviewing the current status of the listed species, the environmental baseline within the action area, the effects of the proposed action, and cumulative effects, it is NMFS' biological opinion that the proposed action is not likely to jeopardize the continued existence of Puget Sound steelhead. No critical habitat has been designated or proposed for this species', so no conclusion is offered with respect its modification.

After reviewing the current status of the listed species, the environmental baseline within the action area, the effects of the proposed action, and cumulative effects, it is NMFS' biological opinion that the proposed action is not likely to jeopardize the continued existence of Puget Sound yelloweye, canary and bocaccio rockfish. No critical habitat has been designated or proposed for these species', therefore none will be affected.

After reviewing the current status of the listed species, the environmental baseline within the action area, the effects of the proposed action, and cumulative effects, it is NMFS' biological opinion that the proposed action is not likely to jeopardize the continued existence of Southern Resident killer whales or to destroy or adversely modify its designated critical habitat.

2.8. Incidental Take Statement

Section 9 of the ESA and Federal regulation pursuant to section 4(d) of the ESA prohibit the take of endangered and threatened species, respectively, without a special exemption. Take is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or to attempt to engage in any such conduct. Harm is further defined by regulation to include significant habitat modification or degradation that results in death or injury to listed species by significantly impairing essential behavioral patterns, including breeding, feeding, or sheltering. Incidental take is defined as take that is incidental to, and not the purpose of, the carrying out of an otherwise lawful activity. For purposes of this consultation, we interpret "harass" to mean an intentional or negligent action that has the potential to injure an animal or disrupt its normal behaviors to a point where such behaviors are abandoned or significantly altered.[6] Section 7(b)(4) and Section 7(o)(2) provide that taking, which is incidental to an otherwise lawful agency action is not considered to be prohibited taking under the ESA, if that action is performed in compliance with the terms and conditions of this incidental take statement.

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[6]NMFS has not adopted a regulatory definition of harassment under the ESA. The World English Dictionary defines harass as "to trouble, torment, or confuse by continual persistent attacks, questions, etc." The U.S. Fish and Wildlife Service defines "harass" in its regulations as an intentional or negligent act or omission which creates the likelihood of injury to wildlife by annoying it to such an extent as to significantly disrupt normal behavioral patterns which include, but are not limited to, breeding, feeding, or sheltering (50 CFR 17.3). The interpretation we adopt in this consultation is consistent with our understanding of the dictionary definition of harass and is consistent with the U.S. Fish and Wildlife interpretation of the term.

2.8.1 Amount or Extent of Take

Puget Sound Chinook Salmon, Steelhead and Rockfish

Individual fish will be present in the action area and experience effects of the proposed action. Upon exposure to effects of the action, fish responses are expected to range from changed normal behavior to death. Therefore incidental take is reasonably certain to occur. The effects of effluent discharged at the Solo Point outfall will injure or kill a small percentage of the ESA-listed fish exposed to chemical constituents in the action area. It is expected that additional injury and mortality will be associated with avoidance behavior that will force fish into deeper water and lead to increased predation ("avoidance/predation"). The NMFS also anticipates that effects of bioaccumulative chemicals present in the discharge will harm some ESA-listed fish when exposed to contaminated water, sediments and prey.

Take Associated with Chemical Exposure and Avoidance Behavior/Predation. As explained in the Opinion, the estimated numbers of juvenile Puget Sound Chinook salmon exposed to and directly affected by the acute mixing zone are 10,530. The estimated numbers of juvenile Puget Sound Chinook salmon directly affected by avoiding the mixing zone and being subjected to increased predation are 21,060. Therefore, estimated take for these effects is 31,590 fish. Our analysis also estimates that 200 juvenile Puget Sound steelhead, 32,620 yelloweye rockfish larvae, 48,930 canary rockfish larvae and 1,060 bocaccio larvae will suffer injury or death when exposed to the acute mixing zone during the same 5-year period.

While it is possible to quantify take in numbers of animals for take associated with chemical exposure and avoidance/predation, it is not possible to measure the number of fish taken. So, instead of quanitifying take in numbers of injured or killed fish, NMFS quantified take the extent of take of these animals based on the best available, observable thresholds which are the monitored concentrations of constituents with well-established biological effects thresholds in fish for aquatic exposure. Specifically, the extent of take indicators for take associated with chemical exposure and avoidance/predation are the residual chlorine and TPH concentrations must not exceed the permit limit; the monitored concentrations of dissolved copper at the outfall, must not exceed 2 µg/L above background at the edge of the chronic mixing zone as modeled after dilution; and the monitored concentrations of dissolved zinc at the outfall, as modeled, must not exceed 5.6 ug/L over background at the edge of the mixing zone as modeled after dilution.

These metrics of the extent of take are good indicators of the take associated with chemical exposure and avoidance/predation because they tend to be representative of wastewater pollutants with recognized toxicological effects to fish. Best available science indicates that as these concentrations increase greater injury and mortality occur, and beyond the identified thresholds, unacceptable levels of take occur. If any of these indicators of the extent of take are exceeded, reinitiation would be warranted.

Take Associated with Bioconcentration and Bioaccumulation. In addition to the take pathways identified above, NMFS anticipates take associated with the bioaccumulative impacts of certain chemicals in the proposed discharge. As explained in the Opinion, this form of take is estimated to be 5,265 juvenile Puget Sound Chinook Salmon, 100 Puget Sound steelhead, 16,310 yelloweye rockfish larvae, 24,465 canary rockfish larvae, and 530 bocaccio larvae over the five year period covered by this opinion.

While it is possible to develop numeric estimates for take associated with bioaccumulation it is not possible to measure the number of fish taken. NMFS has determined that the best surrogate is the monitored concentrations of constituents with reported tissue-residue based biological effects thresholds in fish. Specifically, the extent of take indicator for take associated with bioaccumulation of chemical constituents is the monitored concentrations of arsenic, copper, lead, mercury, cadmium, chromium, and selenium in the Solo Point discharge must not exceed the levels identified in Jarvinen and Ankley (1999) as exceeding tissue-residue based effects in fish when applying their metal-specific bioconcentration factor (for details of these levels, see biological assessment addendum submitted to NMFS on June 3, 2010).

This take surrogate is a good indicator of the take associated with bioconcentration and bioaccumulation of these constituents because they are representative of wastewater pollutants expected to be discharged from Solo Point with recognized potential to bioconcentrate, and for which tissue-residue based effects from bioaccumulation have been reported. Best available science indicates that as these concentrations increase greater injury and mortality occur, and beyond the identified thresholds, unacceptable levels of take are likely. If concentrations of any of the monitored metals measured at the outfall exceed tissue-residue based effects concentrations, based on the reported bioconcentration factors of Jarvinen and Ankley (1999), reinitiation would be warranted.

While there are recognized adverse effects possible from additional constituents in the discharge such as PBDEs, and PPCPs, because tissue-residue thresholds are not fully available, these cannot be used as a extent of take indicator – and the metals identified above represent the best available proxy. NMFS is satisfied with the commitments of the EPA to monitor these additional constituents and work together to resolve appropriate threshold concentrations and develop adaptive management measures as necessary should new information identify additional risk to ESA-listed salmonids or rockfish that has not been considered, per conditions outlined in Section 2.10.

Southern Resident Killer Whales

The increased loading of PBDEs that would occur under the proposed action could result in some level of harm to Southern Resident killer whales by reducing prey quality and increasing PBDE accumulation in the whales. NMFS anticipates that increased PBDE loading in the whales would incur adverse health effects over a shorter period than would otherwise occur absent the action. All individuals of the Southern Resident killer whale DPS have the potential to be adversely affected. However, calves and juveniles are most at risk to harm from the action because they are exposed to PBDEs during developmental growth. Exposure of PBDEs during developmental growth can enhance toxicity to the whales. The extent of take from this adverse impact is not anticipated to increase the risk of mortality for whales currently in the population during the 5-year permit cycle (i.e., and therefore will not rise to the level of serious injury or mortality). The extent of take is likely to be an incrementally small but measurable increase in PBDE concentrations that could cause developmental effects or small reductions in reproductive success or immune function for some individuals, based on our analysis of the whales' PBDE accumulation. Take is within the extent of effects analyzed where the PBDE concentration in the effluent, and the average monthly flow rate (in mgd), are within the range evaluated in this biological opinion (0.165 kg/yr and 7 mgd, respectively). The resulting accumulation of PBDEs in the whales is expected to be 19 and 15 ng/g lipid weight in the adult male and male calf, respectively.

2.8.2 The Effect of Take

The amount of incidental take is based on the requirement that the Solo Point discharge is conducted as described in this opinion. The total amount of incidental take, over the five year period covered by this opinion, includes: 36,855 juvenile (190 adult equivalents) Puget Sound Chinook salmon, 300 juvenile (one adult equivalent) Puget Sound steelhead, 48,930 yelloweye rockfish larvae (0.36 to 0.82 percent of what one adult can produce over a 5-year period), 73,395 canary rockfish larvae (0.77 to 5.8 percent of what one adult can produce over a 5-year period), and 1,590 bocaccio larvae (0.01 to 1.6 percent of what one adult can produce over a 5-year period) and will be measured by the effluent limitations established in the new NPDES permit. The expected extent of take is also a threshold for reinitiating consultation. The project itself does not jeopardize the continued existence of Puget Sound Chinook salmon, Puget Sound steelhead or Puget Sound/Georgia Basin yelloweye rockfish, canary rockfish and bocaccio. But, during the time frame for this permit there will be a diminished capacity for juvenile to adult survival (carrying capacity) for these species.

Given the relatively low numbers calculated for immediate and indirect take of individual fish anticipated from the action, viability characteristics of the population will not be altered. The effect of the take is presumed to delay any improvement of population characteristics in terms of abundance and productivity.

Based on the analyses provided in this biological opinion, NMFS has determined that the level of anticipated incidental take is not likely to jeopardize the continued existence of Puget Sound Chinook salmon, Puget Sound steelhead, Puget Sound/Georgia Basin yelloweye rockfish, canary rockfish, bocaccio, or Southern Resident killer whales.

2.8.3 Reasonable and Prudent Measures and Terms and Conditions

"Reasonable and prudent measures" are nondiscretionary measures to minimize the amount or extent of incidental take (50 CFR 402.02). "Terms and conditions" implement the reasonable and prudent measures (50 CFR 402.14). These must be carried out for the exemption in section 7(o)(2) to apply.

The following reasonable and prudent measures are necessary and appropriate to minimize the impact of incidental take for fish considered in this opinion. The EPA shall:

1) Minimize incidental take from exposure to constituents found in the Solo Point Wastewater Treatment Plant effluent being discharged into Puget Sound.

The terms and conditions described below are non-discretionary, and must be undertaken by the EPA or, if an applicant is involved, must become binding conditions of any permit or grant issued to the applicant, for the exemption in section 7(o)(2) to apply. The EPA has a continuing duty to regulate the activity covered by this incidental take statement. If the EPA: (A) fails to assume and implement the terms and conditions; or (B) fails to require an applicant to adhere to the terms and conditions of the incidental take statement through enforceable terms that are added to the permit or grant document, the protective coverage of section 7(o)(2) may lapse. To monitor the impact of incidental take, the EPA or applicant must report the progress of the action and its impact on the species to NMFS as specified in the ITS.

In order to be exempt from the prohibitions of section 9 of the ESA and implement the reasonable and prudent measures described above, EPA must comply with the following terms and conditions:

1) Provide NMFS with the PBDE and other effluent constituent monitoring reports, including average flow rates, to ensure that the expected PBDE concentrations in the effluent and consequential PBDE loading into Puget Sound falls within the range and extent of take specified in the Incidental Take Statement.

2) Implement all conservation measures that are part of the proposed action in a timely manner, as specified by the time line for completion in the conservation measures included in the proposed action.

All data/reports shall be sent to National Marine Fisheries Service, Washington State Habitat Office, Southwest Washington Branch, Attention: Tim Rymer, 510 Desmond Drive SE, Suite 103, Lacey, Washington 98503-1263. Include the NMFS Tracking Number: 2009/03531 on the report. Included shall be an explanation of why any terms and conditions or minimization measures were not met (if applicable). Project identification shall be provided on all data/reports that includes the following information; project name, type of activity, project location, EPA contact person, and starting and ending dates for data collection/work completed.

2.9. Conservation Recommendations

Section 7(a) (1) of the ESA directs Federal agencies to use their authorities to further the purposes of the ESA by carrying out conservation programs for the benefit of the threatened and endangered species. Specifically, conservation recommendations are suggestions regarding discretionary measures to minimize or avoid adverse effects of a proposed action on listed species or critical habitat or regarding the development of information (50 CFR 402.02).

The following recommendations are discretionary measures that NMFS believes are consistent with this obligation and therefore should be carried out by the EPA. We provide the following conservation recommendations for the conservation of ESA-listed species.

1) Encourage JBLM to initiate the use of ultra-violet disinfection in place of chlorine disinfection.

2) Encourage JBLM to use state of the art treatment that would remove nitrogen, phosphorus, PBDEs and PPCPs to the greatest extent possible.

3) Encourage JBLM to relocate the existing outfall to deeper water as part of the proposed upgrade of the facility.

4) Support the development of marine mammal and fish criteria values for PPCPs and PBDEs at a congener-level.

5) Support the development of programmatic consultation wastewater permits in Puget Sound, where possible.

6) Encourage JBLM to monitor PBDE levels in sediment within and beyond the mixing zone.

7) Encourage JBLM to monitor heavy metal and phthalates within and beyond the mixing zone.

8) Identify ways to work with NMFS through the existing MOA between the Services to make progress on the above conservation measures.

2.10 Reinitiation of Consultation

As provided in 50 CFR 402.16, reinitiation of formal consultation is required where discretionary Federal agency involvement or control over the action has been retained (or is authorized by law) and if: (1) the amount or extent of incidental take is exceeded, (2) new information reveals effects of the agency action on listed species or designated critical habitat in a manner or to an extent not considered in this opinion, (3) the agency action is subsequently modified in a manner that causes an effect on the listed species or critical habitat not considered in this opinion, or 4) a new species is listed or critical habitat designated that may be affected by the action.

To reinitiate consultation, contact the Washington State Habitat Office of NMFS, and refer to the NMFS Tracking Number assigned to this consultation (2009/03531).

Please notify NMFS if the EPA carries out any of the above recommendations so that we will be kept informed of actions that are intended to improve the conservation of listed species or their designated critical habitats.

2.11 Not Likely to Adversely Affect Determinations

Steller Sea Lions and Humpback Whales

Steller sea lions in Washington are from the eastern DPS. For the past 20 to 30 years, the eastern DPS has grown steadily at about 3 percent per year. In the final revised recovery plan (73 FR 11872, NMFS 2008) no threats to the continued recovery of the eastern DPS were identified. Nevertheless, NMFS evaluates whether the proposed action has the potential to affect Steller sea lions. Steller sea lions can occur in Washington waters throughout the year, however there are no breeding rookeries in Washington. Occurrence in inland waters of Washington is limited to primarily male and sub-adult Steller sea lions in fall, winter, and spring months. Steller sea lions use haulout locations in coastal and inland waters of Washington, including in Puget Sound. The nearest consistently used haulout locations to the Solo Point wastewater outfall are located at the Toliva Shoals buoy, approximately 5 miles north where as many as 10 individuals have been observed hauled-out on the buoy, and the north side of the Nisqually River delta, approximately 3 miles south where as many as 100 individuals have been observed hauled-out. Steller sea lions that use these haulouts may forage in the vicinity of the mixing zone. They are generalist predators that eat a variety of fishes and cephalopods.

Humpback whales of the eastern North Pacific stock occur in coastal waters off the U.S. west coast, including waters of Washington State. The stock feeds off the U.S. west coast, with winter migratory destination in coastal waters of Mexico and Central America. In recent years, humpback whales are sighted with increasing frequency in the inside waters of Washington, including Puget Sound (primarily during the fall and spring); however, occurrence is uncommon. Humpback whales more commonly occur in coastal waters and forage on a variety of crustaceans, other invertebrates, and forage fish.

The proposed action may affect the quality of prey for these marine mammal species by introducing contaminants into the aquatic food web. PBDEs, pharmaceuticals, metals, and other toxic chemicals are found in effluent from wastewater treatment plants and can persist in the environment. Phytoplankton, zooplankton, benthic invertebrates, demersal fish, forage fish, and other fishes can be exposed to and ingest these contaminants found in wastewater effluent. As these exposed organisms are consumed, the contaminants can biomagnify up the food chain and can accumulate in upper-trophic level species. Our analysis examines the extent to which any potential food web transfer of these contaminants may result. This would determine the extent to which the proposed action could result in bioaccumulation in these marine mammal species.

Steller sea lions are likely to occur in the vicinity of the mixing zone. It is also likely that Steller sea lions will be exposed to pollutants from the discharged effluent through ingestion of prey, however, the extent of exposure is largely unknown. Unlike Southern Resident killer whales that consume primarily salmonids, a highly contaminated upper-trophic level prey, Steller sea lions have a large foraging base and consume prey at lower trophic levels. Steller sea lions are, therefore, likely exposed to less contaminated prey than the Southern Resident killer whales. There is limited information on the contaminant levels in Steller sea lions. Overall, the studies suggest a decline in contaminant concentrations over time, which is consistent with that reported for other wildlife species (NMFS 2008). Heavy metal concentrations in Steller sea lions are generally lower than northern fur seals (Noda et al. 1995; Beckmen et al. 2002). Additionally, comparable levels of zinc, copper, and metallothionein were measured in pups from both the eastern and western Steller sea lion DPSs (Castellini 1999). Although these studies are not comprehensive, they indicate that heavy metals were not likely a significant factor in the decline of the Steller sea lions (NMFS 2008). Currently, no studies have reported PBDE concentrations in Steller sea lions and the risks associated are largely unknown. However, the population has grown steadily for the past 20 to 30 years with no indication that contaminant-induced health effects are limiting recovery. Exposure of a small number of sea lions to a variety of lower trophic level prey in the mixing zone for part of the year is not likely to increase contaminant concentrations in individuals to harmful levels. For these reasons, the potential for exposure from ingesting contaminated prey and any subsequent chance of bioaccumulation in Steller sea lions is anticipated to be insignificant. The proposed action may reduce the quantity of salmonid prey available, as described in the incidental take statement. NMFS anticipates similar effects on non-listed fishes. As described above, the extent of anticipated ESA-listed Chinook salmon take is likely limited in amount and is estimated to be less than 100 adult equivalents per year. Any salmonid take up to the aforementioned maximum extent and amount would result in an insignificant reduction in prey resources for Steller sea lions that may intercept these species within their range.

Humpback whales are extremely unlikely to be present or consume prey in the inland waters of Puget Sound. Thus, any potential for exposure from ingesting contaminated prey and any subsequent chance of bioaccumulation in humpback whales is extremely unlikely and therefore discountable.

Because all potential adverse effects to Steller sea lions and humpback whales are insignificant or discountable, NMFS has determined that the proposed action “may affect, but is not likely to adversely affect” Steller sea lions and humpback whales.

3. MAGNUSON-STEVENS FISHERY CONSERVATION AND MANAGEMENT ACT ESSENTIAL FISH HABITAT CONSULTATION

The consultation requirement of section 305(b) of the MSA directs Federal agencies to consult with NMFS on all actions or proposed actions that may adversely affect EFH. The MSA (section 3) defines EFH as "those waters and substrate necessary to fish for spawning, breeding, feeding, or growth to maturity." Adverse effects include the direct or indirect physical, chemical, or biological alterations of the waters or substrate and loss of, or injury to, benthic organisms, prey species and their habitat, and other ecosystem components, if such modifications reduce the quality or quantity of EFH. Adverse effects on EFH may result from actions occurring within EFH or outside EFH, and may include site-specific or EFH-wide impacts, including individual, cumulative, or synergistic consequences of actions (50 CFR 600.810). Section 305(b) also requires NMFS to recommend measures that can be taken by the action agency to conserve EFH.

3.1 Essential Fish Habitat Affected by the Project

This analysis is based, in part, on the EFH assessment provided by the EPA and descriptions of EFH for Pacific coast salmon (PFMC 1999) contained in the fishery management plans developed by the Pacific Fishery Management Council (PFMC) and approved by the Secretary of Commerce. The action area includes locations designated as EFH for various life-history stages of Chinook salmon, coho salmon, Puget Sound pink salmon, 46 species of groundfish, and 4 coastal pelagic species (see Table 18). Further, Puget Sound is a Habitat Area of Particular Concern (HAPC), based on importance of the ecological function provided by the habitat. The environmental effects of the proposed project may adversely affect up to approximately 7.5 acres of EFH in the HAPC for these species.

Table 19: Species with designated EFH found in waters of Puget Sound

Groundfish Species	Redstripe rockfish *(Sebastes proriger)*	Dover sole *(Microstomus pacificus)*
Spiny dogfish *(Squalus acanthias)*	Rosethorn rockfish *(S . helvomaculatus)*	English sole *(Parophrys vetulus)*
Big skate *(Raja binoculata)*	Rosy rockfish *(S. rosaceus)*	Flathead sole *(Hippoglassoides elassodon)*
California skate *(R. inornata)*	Rougheye rockfish *(S. aleutianus)*	Petrale sole *(Eopsetta jordani)*
Longnose skate *(R. rhina)*	Sharpchin rockfish *(S. zacentrus)*	Rex sole *(Glyptocephalus zachirus)*
Ratfish *(Hydrolagus colliei)*	Splitnose rockfish *(S. diploproa)*	Rock sole *(Lepidopsetta bilineata)*

Pacific cod (*Gadus macrocephalus)*	Stripetail rockfish *(S. saxicola)*	Sand sole *(Psettichthys melanostictus)*
Pacific whiting (Hake) *(Merluccius productus)*	Tiger rockfish *(S. nigrocinctus)*	Starry flounder *(Platyichthys stellatus)*
Black rockfish *(S. melanops)*	Vermillion rockfish *(S. miniatus)*	Arrowtooth flounder *(Atheresthes stomias)*
Bocaccio *(S. paucispinis)*	Yelloweye rockfish *(S. ruberrimus)*	**Coastal Pelagic Species**
Brown rockfish *(S. auriculatus)*	Yellowtail rockfish *(S. flavidus)*	Northern anchovy *(Engraulis mordax)*
Canary rockfish *(S. pinniger)*	Shortspine thornyhead *(Sebastolobus alascanus)*	Pacific sardine *(Sardinops sagax)*
China rockfish *(S. nebulosus)*	Cabezon *(Scorpaenichthys marmoratus)*	Pacific mackerel *(Scomber japonicus)*
Copper rockfish *(S. caurinus)*	Lingcod *(Ophiodon elongatus)*	Market squid *(Loligo opalescens)*
Darkblotched rockfish *(S. crameri)*	Kelp greenling *(Hexagrammos decagrammus)*	**Salmon**
Greenstriped rockfish *(S. elongatus)*	Sablefish *(Anoplopoma fimbria)*	Chinook salmon *(Oncorhynchus tshawytscha)*
Pacific Ocean perch *(S. alutus)*	Pacific sanddab *(Citharichthys sordidus)*	Coho salmon *(O. kisutch)*
Quillback rockfish *(S. maliger)*	Butter sole *(Isopsetta isolepsis)*	Pink Salmon *(O. gorbuscha)*
Redbanded rockfish *(S. babcocki)*	Curlfin sole *(Pleuronichthys decurrens)*	

3.2 Adverse Effects on Essential Fish Habitat

The NMFS determined that the proposed action will have adverse effects to EFH designated for Puget Sound Chinook salmon, Puget Sound/Georgia Basin yelloweye rockfish, canary rockfish and bocaccio. This was based on information provided in the BE and the analysis of effects presented in the ESA portion of this document (Section 2.4). Puget Sound Chinook salmon are an ESA and MSA-managed species. The other MSA-managed salmon, Puget Sound coho and Puget Sound pink salmon, and associated EFH will experience very similar effects as those discussed in the ESA Section 2.4 for Chinook salmon. Juvenile and adult Chinook, coho and pink salmon are known to make use of the action area (Fresh et al. 1979; Duffy 2003). Adults stage in and migrate through it en route to rivers/streams where they travel to spawning areas. Juveniles rear in and emigrate through the action area, utilizing the nearshore for cover, refuge and forage.

There is also use by larvae, young of the year, juvenile and adult ground fish and coastal pelagic fish species listed in Table 19 (Fresh et al. 1979; Palsson et al. 2009). The life history stages for these fish would be spawning and rearing in this portion of South Puget Sound. Given the effluent from Solo Point WWTP discharges 24 hours/day, 7days/week, and 365 days/year, water quality will be degraded as will habitat within the action area for as long as the discharge continues. Adverse effects include the direct/indirect physical, chemical, and biological alterations of the water and/or substrate. Impacts also encompass loss of, or injury to, benthic organisms, prey species and their habitat, as well as other ecosystem components. These effects will be similar to those described in the ESA (Section 2.4) portion of this document.

3.3 Essential Fish Habitat Conservation Recommendations

The following conservation measures are necessary to avoid, mitigate, or offset the impact of the proposed action on EFH:

1) As soon as possible, initiate the use of ultra-violet disinfection in place of chlorine disinfection.

2) Utilize state of the art treatment that would remove nitrogen, phosphorus, PBDEs and PPCPs to the greatest extent possible.

3) Relocate the existing outfall to deeper water as part of the proposed upgrade of the facility.

4) Conduct sediment sampling within and beyond the mixing zone for heavy metals, phthalates, and PBDEs.

5) Begin developing criteria/threshold values for PPCPs and PBDEs (at a congener-level).

6) Consult programmatically on all non-federal WWTP outfalls in Puget Sound.

NMFS expects that full implementation of these EFH conservation recommendations would protect, by avoiding or minimizing the adverse effects described in section 3.2 above, approximately 7.5 acres of designated EFH for Pacific coast salmon, Pacific coast groundfish, and coastal pelagic species.

3.4 Statutory Response Requirement

As required by section 305(b) (4) (B) of the MSA, the Federal agency must provide a detailed response in writing to NMFS within 30 days after receiving an EFH Conservation Recommendation from NMFS. Such a response must be provided at least 10 days prior to final approval of the action if the response is inconsistent with any of NMFS' EFH Conservation Recommendations, unless NMFS and the Federal agency have agreed to use alternative time frames for the Federal agency response. The response must include a description of measures proposed by the agency for avoiding, mitigating, or offsetting the impact of the activity on EFH. In the case of a response that is inconsistent with NMFS Conservation Recommendations, the

Federal agency must explain its reasons for not following the recommendations, including the scientific justification for any disagreements with NMFS over the anticipated effects of the action and the measures needed to avoid, minimize, mitigate, or offset such effects [50 CFR 600.920(k)(1)].

In response to increased oversight of overall EFH program effectiveness by the Office of Management and Budget, NMFS established a quarterly reporting requirement to determine how many conservation recommendations are provided as part of each EFH consultation and how many are adopted by the action agency. Therefore, we ask that in your statutory reply to the EFH portion of this consultation, you clearly identify the number of conservation recommendations accepted.

3.5 Supplemental Consultation

The (Federal action agency) must reinitiate EFH consultation with NMFS if the proposed action is substantially revised in a way that may adversely affect EFH, or if new information becomes available that affects the basis for NMFS' EFH conservation recommendations [50 CFR 600.920(l)].

4. DATA QUALITY ACT DOCUMENTATION AND PRE-DISSEMINATION REVIEW

Section 515 of the Treasury and General Government Appropriations Act of 2001 (Data Quality Act) specifies three components contributing to the quality of a document. They are utility, integrity, and objectivity. This section of the Opinion addresses these Data Quality Act (DQA) components, documents compliance with the DQA, and certifies that this Opinion has undergone pre-dissemination review.

Utility: Utility principally refers to ensuring that the information contained in this consultation is helpful, serviceable, and beneficial to the intended users. The intended users of this Opinion are the EPA and its applicant. This Opinion will be posted on NMFS Northwest Region web site (http://www.nwr.noaa.gov). The format and naming adheres to conventional standards for style.

Integrity: This consultation was completed on a computer system managed by NMFS in accordance with relevant information technology security policies and standards set out in Appendix III, 'Security of Automated Information Resources,' Office of Management and Budget Circular A-130; the Computer Security Act; and the Government Information Security Reform Act.

Objectivity:

Information Product Category: Natural Resource Plan.

Standards: This consultation and supporting documents are clear, concise, complete, and unbiased; and were developed using commonly accepted scientific research methods. They adhere to published standards including the NMFS ESA Consultation Handbook, ESA Regulations, 50 CFR 402.01, et seq., and the MSA implementing regulations regarding EFH, 50 CFR 600.920(j).

Best Available Information: This consultation and supporting documents use the best available information, as referenced in the Literature Cited section. The analyses in this Opinion/EFH consultation contain more background on information sources and quality.

Referencing: All supporting materials, information, data and analyses are properly referenced, consistent with standard scientific referencing style.

Review Process: This consultation was drafted by NMFS staff with training in ESA and MSA implementation, and reviewed in accordance with Northwest Region ESA quality control and assurance processes.

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APPENDIX A. Constituents monitored as part of pretreatment permit requirements, per Table II of Appendix of 40 CFR

Volatiles
1V acrolein
2V acrylonitrile
3V benzene
5V bromoform
6V carbon tetrachloride
7V chlorobenzene
8V chlorodibromomethane
9V chloroethane
10V 2-chloroethylvinyl ether
11V chloroform
12V dichlorobromomethane
14V 1,1-dichloroethane
15V 1,2-dichloroethane
16V 1,1-dichloroethylene
17V 1,2-dichloropropane
18V 1,3-dichloropropylene
19V ethylbenzene
20V methyl bromide
21V methyl chloride
22V methylene chloride
23V 1,1,2,2-tetrachloroethane
24V tetrachloroethylene
25V toluene
26V 1,2-trans-dichloroethylene
27V 1,1,1-trichloroethane
28V 1,1,2-trichloroethane
29V trichloroethylene
31V vinyl chloride
Acid Compounds
1A 2-chlorophenol
2A 2,4-dichlorophenol
3A 2,4-dimethylphenol
4A 4,6-dinitro-o-cresol
5A 2,4-dinitrophenol
6A 2-nitrophenol

7A 4-nitrophenol
8A p-chloro-m-cresol
9A pentachlorophenol
10A phenol
11A 2,4,6-trichlorophenol
Base/Neutral
1B acenaphthene
2B acenaphthylene
3B anthracene
4B benzidine
5B benzo(a)anthracene
6B benzo(a)pyrene
7B 3,4-benzofluoranthene
8B benzo(ghi)perylene
9B benzo(k)fluoranthene
10B bis(2-chloroethoxy)methane
11B bis(2-chloroethyl)ether
12B bis(2-chloroisopropyl)ether
13B bis (2-ethylhexyl)phthalate
14B 4-bromophenyl phenyl ether
15B butylbenzyl phthalate
16B 2-chloronaphthalene
17B 4-chlorophenyl phenyl ether
18B chrysene
19B dibenzo(a,h)anthracene
20B 1,2-dichlorobenzene
21B 1,3-dichlorobenzene
22B 1,4-dichlorobenzene
23B 3,3′-dichlorobenzidine
24B diethyl phthalate
25B dimethyl phthalate
26B di-n-butyl phthalate
27B 2,4-dinitrotoluene
28B 2,6-dinitrotoluene
29B di-n-octyl phthalate
30B 1,2-diphenylhydrazine (as azobenzene)
31B fluroranthene
32B fluorene
33B hexachlorobenzene
34B hexachlorobutadiene
35B hexachlorocyclopentadiene
36B hexachloroethane
37B indeno(1,2,3-cd)pyrene
38B isophorone
39B napthalene
40B nitrobenzene

41B N-nitrosodimethylamine
42B N-nitrosodi-n-propylamine
43B N-nitrosodiphenylamine
44B phenanthrene
45B pyrene
46B 1,2,4-trichlorobenzene
Pesticides
1P aldrin
2P alpha-BHC
3P beta-BHC
4P gamma-BHC
5P delta-BHC
6P chlordane
7P 4,4′-DDT
8P 4,4′-DDE
9P 4,4′-DDD
10P dieldrin
11P alpha-endosulfan
12P beta-endosulfan
13P endosulfan sulfate
14P endrin
15P endrin aldehyde
16P heptachlor
17P heptachlor epoxide
18P PCB-1242
19P PCB-1254
20P PCB-1221
21P PCB-1232
22P PCB-1248
23P PCB-1260
24P PCB-1016
25P toxaphene